United States
Department of
Agriculture

Forest Service

**Rocky Mountain
Forest and Range
Experiment Station**

Fort Collins
Colorado 80526

**General Technical
Report RM-248**

Effect of Management on Water Quality in North American Forests

Thomas C. Brown and Dan Binkley

Acknowledgment

The authors are grateful to Lee MacDonald and Gordon Stuart for helpful reviews.

This publication was printed on recycled paper.

USDA Forest Service
General Technical Report RM-248

June 1994

Effect of Management on Water Quality in North American Forests

Thomas C. Brown
Rocky Mountain Forest and Range Experiment Station[1]

Dan Binkley
Colorado State University

[1]*Headquarters is in Fort Collins, in cooperation with Colorado State University*

Contents

Page

Introduction 1
Pathogens 3
  Background 3
  Effects of Management 4
Dissolved Oxygen 4
  Background 4
  Effects of Management 5
Nutrients 5
  Background 5
  Effects of Management 6
Dissolved Solids 10
  Background 10
  Effects of Management 11
Sediments 11
  Background 11
  Effects of Management 13
Toxics 15
  Background 15
  Effects of Management 15
Water Temperature 16
  Background 16
  Effects of Management 16
Water Quality Protection Programs 18
  Type of Program 18
  Financial Incentives 19
  Implementation Monitoring 19
  Effectiveness Monitoring 20
  Federal Land Management Agency Nonpoint Source Pollution Control Efforts 20
  Cost Effectiveness of BMPs 20
Summary and Conclusions 21
References 22

Effect of Management on Water Quality in North American Forests

Thomas C. Brown and Dan Binkley

Introduction

Since passage of the Federal Water Pollution Control Act amendments in 1972, considerable progress has been made in reducing municipal and industrial releases of water pollutants. Less progress has been made in controlling water pollutants that reach the Nation's rivers and streams in runoff. Land and their uses are sources of many potential water quality problems.

"Water quality" is affected by a series of physical (e.g., suspended sediment), chemical (e.g., nitrate, pesticides), and biological (e.g., giardia) components or indicators. The levels of these components can be affected greatly by both natural events and human actions. For example, the level of suspended sediment can increase greatly after events such as wildfires or timber harvesting. Human actions also can lower some concentrations below the natural level, as prolonged fire protection has done in many forest areas.

Water quality can degrade because of natural events or human-caused (i.e., management-related) activities. It is more likely to occur after human-caused influences if the natural concentration is already high. Human-caused sources of water quality degradation are broadly categorized as "point source" or "nonpoint source." Effluents causing point source pollution leave the source in a contained structure,such as a pipe or canal. The water quality impacts of these effluents can be monitored, providing the manager with direct information about the source and indicating if corrective action is needed. Sewage treatment plants and industrial facilities are important point sources. Nonpoint source pollution occurs as more diffuse runoff from land areas. The dispersed transport mechanisms of runoff make it difficult to monitor the nonpoint source degradation of water quality, except at points downstream from the cause, where it often is difficult to identify the specific land area from which the degradation originated or the activity that caused it.

"Pollution" typically refers to water quality degradation caused by human influences. The 1972 Act essentially defined pollution as any alteration in water quality caused by human use of resources. It also called for sufficient control of all pollution sources so that waters were fishable and swimmable by 1983, and elimination of all point source pollution discharges by 1985. These goals were not met; some minimum level of pollution is physically or economically unavoidable because of the limitations of available technology (National Water Commission 1973, Black 1992). Water quality protection now emphasizes the interim, nationwide fishable and swimmable goals established in the 1972 Act, as well as support of "designated" water uses determined by the states (EPA 1990). For some uses, the levels at which some components or indicators become harmful have been agreed upon (e.g., water drinking standards), while for other uses or components (e.g., nutrients) there is no consensus about the maximum acceptable levels of water quality degradation.

Urban areas are sources of all major categories of water pollution. Runoff from urban areas carries household chemical products, pet wastes, yard applications, industrial chemicals, transportation byproducts, construction-displaced sediment, and other wastes to rivers and streams. Farms — the location of soil tillage, fertilizer and pesticide applications, irrigation, and animal concentrations — are also important sources of all major categories of water pollution. Mines, landfills, animal feed lots, and rural septic systems are also common sources of some categories of pollution. Forests and rangelands are sources of nutrients, oxygen-demanding organic material, and suspended sediments, and also can yield toxics, if pesticides are used. Water temperature also can increase in forest areas, after harvesting or fire.

Two national surveys report on sources of water pollution in the U.S. The U.S. Environmental Protection Agency (EPA) biennially publishes a national assessment of water quality, summarizing state reports that are based on monitoring, surveys of scientists, water quality modeling, and citizen input. The 1988 assessment (EPA 1990) reports data on about 25% of the Nation's miles of rivers and streams, that were assessed by about 35 states. The "designated uses" (e.g., contact recreation, drinking water supply, high-quality cold water fishery) were judged to be not fully "supported" by adequate water quality along about 30% of the assessed river miles. Agricultural runoff was the largest source of problems, impairing 20% of the affected river and stream miles (fig. 1). Municipal discharges, mining, and urban runoff were other common sources. Forestry activities (mainly harvesting and related road construction) impaired 3% of the assessed river miles. Effects of rangeland management were not included as a separate category.

The 1982 National Fisheries Survey (Judy et al. 1984) relied on evaluations by state fisheries biologists of a statistically-based sample of 1,303 (approximately 10%) of the Nation's river reaches. The biologists reported that "the survival, productivity, or use of the fish community [was] adversely affected" in 56% of the sampled stream miles (and 45% of the perennial stream miles). This survey also placed agricultural runoff at the top of the list, but ranked forestry activities second, affecting 8% of the river miles (fig. 1). The higher percentages for this survey,

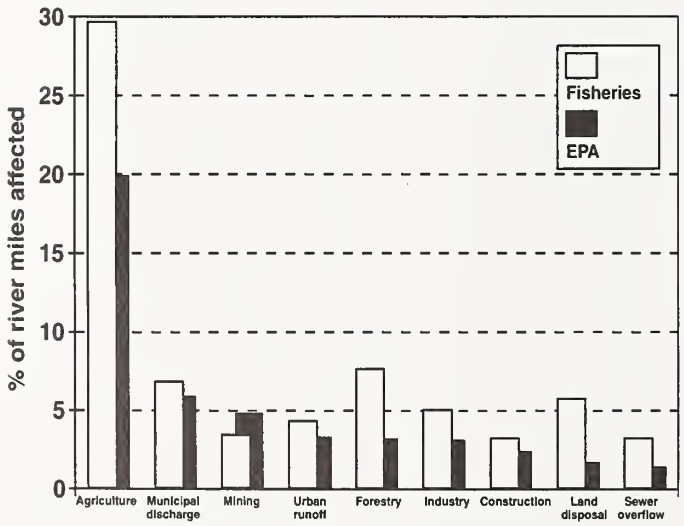

Figure 1.—Incidence of major sources of water pollution according to two national surveys

Table 1.—Major categories of water pollution and related water quality constituents.

Category	Principal constituents or measures
Pathogenic organisms	Bacteria (e.g., fecal coliform, fecal streptococci)
	Protozoa (e.g., giardia)
	Certain viruses and fungi
Organic material	Biochemical oxygen demand (BOD)
	Dissolved oxygen
Nutrients	Nitrogen (nitrate, nitrite, ammonia)
	Phosphorus (dissolved ions and inorganic molecules)
Dissolved solids	Specific ions (e.g., sodium, potassium, calcium, magnesium)
	General measures (total dissolved solids (TDS), conductivity)
Sediments	Suspended sediment (measured as concentration or turbidity)
	Stream bottom sedimentation
Toxics	Metals (e.g, Cadmium, Lead, Mercury)
	Pesticides (insecticides, herbicides, fungicides)
	Other (mainly organics such as Polyclorinated biphenyls (PCBs))
Temperature	Degrees Celsius

compared with the EPA summary, may result from the replacement of a "designated use" criterion with a singular concern about fish habitat, and from different definitions of impairment.

The EPA summary provides only an approximate indication of the incidence of water pollution, because of the variety of methods used by the different states, and because the sample of assessed rivers and lakes was not systematically designed (some states did not report, and some reporting states focused heavily on problem areas). Furthermore, the fisheries survey may overly emphasize fish habitat for some rivers. However, these surveys indicate at least the relative importance of the different categories of water pollution. They clearly indicate that agriculture is the most prevalent nonpoint source of water quality problems, and that forest management activities are important concerns in relatively few locations.

Sources of water quality degradation can be grouped into seven categories (table 1). **Pathogenic organisms** are water-borne disease-causing agents, including certain bacteria (indicated by measures such as fecal coliform and fecal streptococci bacte-

ria), certain protozoa (e.g., giardia), harmful viruses, and certain fungi. This category of water pollution has been adequately controlled, in most urban areas of developed countries, by effective water treatment and distribution procedures.

Organic material, from waste products and decaying plants, requires oxygen as it is decomposed by bacteria. This decay process creates "biochemical oxygen demand," lowering the dissolved oxygen available to fish and aquatic invertebrates, potentially to lethal levels, and also potentially causing water color changes and odor problems.

Nutrients primarily include forms of phosphorus and nitrogen. Nutrients are essential for primary food production in aquatic ecosystems.Howeer, at high levels, they cause excessive growth of aquatic plants and animals, which, in turn, causes murky water, floating algae, and dense mats of aquatic plants — a condition known as "eutrophication" — that restricts water use for recreation, fish, and wildlife. As this organic matter decays, it also may reduce dissolved oxygen. In addition, elemental phosphorus and nitrogen (as nitrate, nitrite, and ammonia) can be directly toxic to fish and humans (nitrate turns into nitrite in the body, which can be especially toxic to infants).

Dissolved solids include a series of ions, commonly called salts, as well as dissolved organic compounds. Total dissolved solids (TDS) and specific conductivity are two measures of the overall concentration of these ions. High concentrations of dissolved solids corrode pipes and water-using appliances, reduce yields of some irrigated crops, require increased use of soaps and detergents, and can harm fish and other aquatic organisms.

Sediments consist of fine soil particles that are carried along in stream flow, and heavier particles that settle on the stream bottom. Suspended sediments increase turbidity and transport plant nutrients, heavy metals, pesticides, pathogens, and other potential pollutants attached to the soil particles. Sustained high turbidity can reduce photosynthesis by algae, reduce the success of sight-feeding fish, and perhaps alter food chains. High suspended sediment concentrations also degrade the quality of drinking water. Settling soil particles reduce the porosity of gravel beds, generating anaerobic conditions unsuitable for spawning and blocking emergence of fry from the gravels. Settling soil particles also reduce water storage capacity of reservoirs and obstruct navigation.

"Water quality" may be perceived to include or exclude the condition of stream channels. This report does not include discussion of stream bottom sediments and other stream channel issues, although it is recognized that settling of suspended sediment affects channel condition. MacDonald et al. (1991) provide more information on stream channel condition as related to water quality.

Toxics are chemicals that can cause adverse effects at extremely low concentrations. They include toxic heavy metals (e.g., mercury, lead, cadmium, arsenic) and other synthetic, generally organic, pesticides and industrial materials (e.g., PCBs). There are more than 60,000 commercial chemical substances in use in the U.S., and the impact of most in aquatic environments is unknown.

Water **temperature** affects both chemical and biological characteristics of streams. For example, the solubility of oxygen decreases rapidly as temperature increases; and most aquatic organisms have optimal temperature ranges. Some fish species require a narrow range of temperature, while others are more tolerant of temperature changes.

The following sections review what has been learned in the United States (and, to some extent, in Canada) about the effects of forest management on each of these seven categories of water quality, and what is happening to mitigate those effects.

Pathogens

Background

Most interest in microbiological water quality centers on organisms that are pathological to humans, or on generalist bacteria (such as total coliform counts) that may be overall indicators of microbial contamination (MacDonald et al. 1991). Total coliform counts have been used widely to assess drinking water quality. Two more specific classes of bacteria are commonly examined: fecal coliform (mostly from feces of humans and other mammals), and fecal streptococci (usually from mammals other than humans). The ratio of fecal coliform to fecal streptococci is used sometimes to differentiate between human and animal sources of pollution, although its utility in distinguishing the source has been questioned (EPA 1978).

Another water borne disease, *Giardia lambila* (a flagellated protozoan), is an important water quality concern in many western mountains (Brown 1989). Many cases of *Giardia* can be traced to streams with substantial beaver activity; but the role of other mammals in spreading the disease is not clear. Other specific pathogens also have been found in forest water supplies. For example, in several California national forests, the bacterium cryptosporidium was detected in campground water systems. The incidence of this bacterium may be related to drought conditions, as well as to increasing rigor of inspection required by new water quality laws.

Lettenmaier et al. (1991) found that 15-20% of the U.S. Geological Survey NASQAN stations that they studied had significant trends in pathogens, during the period 1978-87 (table 2). Stations with decreases were about twice as prevalent as stations with increases. Earlier, Smith et al. (1987) also found more decreases in fecal bacteria (approximately 18% of the stations decreased, while only 4% increased). Increases in pathogens were associated with larger proportions of the land area in pasture or urban uses. The decreases reflect the increasing success at municipal and rural domestic water treatment.

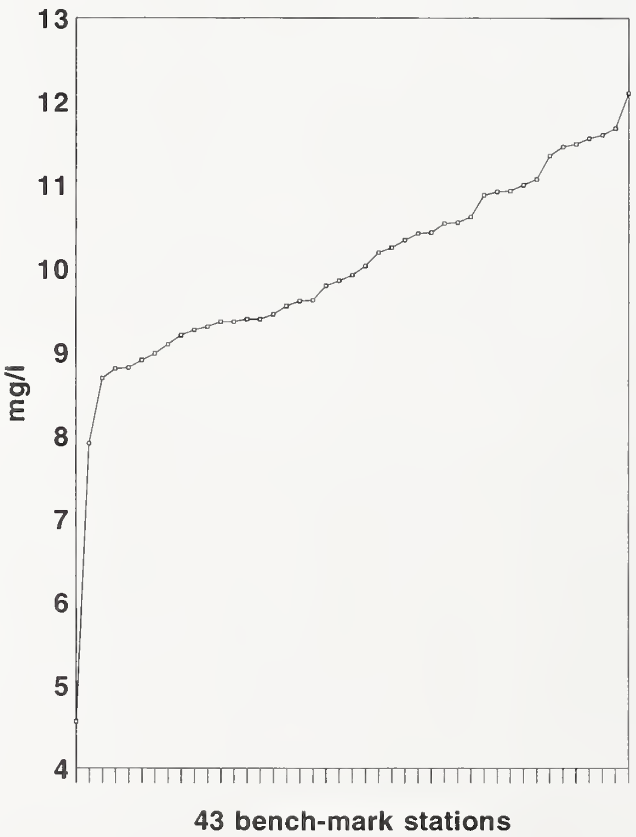

Figure 2.—Mean annual dissolved oxygen concentration at 43 benchmark stations draining areas of forest and rangeland (stations shown in order of increasing dissolved oxygen)

Effects of Management

Pathogens are not affected by timber harvesting. Grazing is the primary land use that may increase microbial contamination in forest streams (Buckhouse and Gifford 1976). Few studies have carefully compared coliform levels in grazed and ungrazed areas. One watershed-level study, in the Bear River Range of northern Utah (Darling and Coltharp 1973, cited in Brown 1989), found maximum total coliform counts of about 150 colonies/100 ml, in an ungrazed watershed, compared with maximums of 700 colonies/100 ml, for a sheep-grazed watershed, and 1,500 colonies/100 ml, for a cattle-grazed watershed. Concentrated recreation use and wildlife populations, as well as mountain homes with inadequate waste disposal systems, also can increase pathogens to unacceptable levels (Potter et al. 1984). Except for grazing impacts, pathogen problems on forest lands that are caused by human activities generally are isolated and are readily solved. For example, recreation impacts can be ameliorated by providing facilities or limiting use rates.

Dissolved Oxygen

Background

Forest streams typically contain 8 to 12 mg of oxygen/l, with lower concentrations in streams with high organic matter and high temperature. We selected 43 U.S. Geological Survey bench-mark stations (Biesecker and Leifeste 1975), in 33 states, that were draining relatively undisturbed areas largely covered with forest or range land vegetation (Binkley and Brown 1993b). At these stations, mean annual dissolved oxygen concentration was 7.9 mg/l or greater for all but one of them (fig. 2). However, mean annual data do not indicate short-term drops in concentrations that may be critical for fish. Streams containing spawning salmonid fish should not drop below 8 mg of O_2/l for one day, or below 9.5 mg/l for a 7-day mean; concentrations of 5 to 6.5 mg/l may be sufficient for adults (MacDonald et al. 1991).

Lettenmaier et al. (1991) found that only 5% of the NASQAN stations had significant trends in oxygen deficit, with decreases three times as common as increases (table 2). The decreases in deficit probably were associated with improvement in point-source discharges.

Effects of Management

Only a few studies have documented depressed oxygen concentrations in streams as a result of forest management activities. In Quebec, Plamondon et al. (1982) examined the effects of massive inputs of logging debris into a low-gradient stream (<1%). The logging debris impounded the stream, and lowered dissolved oxygen concentrations to near 0. In the Coast Range of Oregon, Hall and Lantz (1969) found that accumulation of logging debris in a small stream depressed dissolved oxygen concentrations as low as 3 mg/l in summer (relative to a control stream with about 10 mg/l). The Caspar Creek Watershed Study in California (Rice et al. 1979) recorded depressions in dissolved oxygen concentrations in logged areas to 5 mg/l in summer, compared with an expected atmospheric equilibrium concentration of about 7 mg/l.

Table 2.—Water quality trends 1978-1987.[1]

Constituent	Number of stations[2]	Percent with significant trend[3]	
		positive	negative
Pathogens			
Fecal coliform	390	6	13
Fecal streptococcus	366	5	10
Organic material			
Oxygen deficit	316	4	12
Plant nutrients			
Total nitrogen	390	21	6
Total phosphorus	389	3	18
Dissolved solids			
TDS	388	22	6
Suspended sediment	153	8	12
Toxics			
Arsenic	383	1	25
Cadmium	360	1	16
Lead	374	1	12

[1]*Source: Lettenmaier (1991). Trends in flow-adjusted concentration.*

[2]*U.S. Geological Survey NASQAN data (Ficke and Hawkinson 1975).*

[3]*Percent of the stations with positive and negative trends at the 0.1 significance level.*

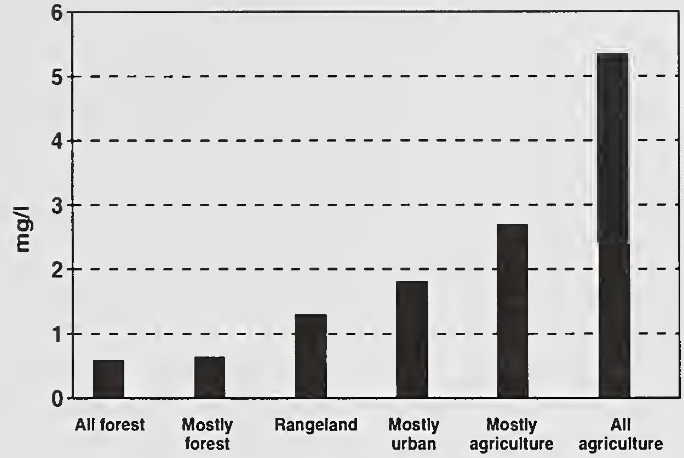

Figure 3.—Nitrogen concentration by land cover (from Omernik 1977)

Very heavy inputs of fine organic debris to low-flow streams can lower dissolved oxygen levels below 1 mg/l (Brown 1989); but overall, the input of fine organic debris after forest harvesting usually is low enough that dissolved oxygen concentrations are not substantially depressed, especially where high turbulence replenishes oxygen from the atmosphere (Ice 1978). Current forest practices generally do not add enough debris to streams to have a substantial effect (MacDonald 1991). However, forest practices that do not lower the concentration of dissolved oxygen in the water column still may lower the oxygen concentration in the streambed gravels, by adding fine sediments that impede inflow of well-aerated water and downward diffusion of oxygen (Everest et al. 1987, MacDonald et al. 1991).

Nutrients

Background

In a nationwide analysis, Omernik (1977) found that annual nutrient concentrations (both total phosphorus and total nitrogen), in streams draining predominately agricultural watersheds, were about nine times higher than in streams draining predominantly forested watersheds, and about four times higher than in streams draining predominantly rangeland watersheds (fig. 3). Percent of the watershed in agricultural and urban uses correlated positively with nutrient concentrations in all regions of the U.S., while percent of the watershed in forest cover correlated negatively with nutrient concentrations in nearly all areas.

In nationwide trends, both Smith et al. (1987) and Lettenmaier et al. (1991) found many stations showing increases in nitrogen and decreases in phosphorus concentrations. The latter study found that 21% of the stations increased in nitrogen concentration, while 6% decreased (table 2). The uptrends were distributed rather evenly over the continental U.S. Smith et al. suggested that changes in agricultural fertilizer use and atmospheric deposition accounted for the significant trends; but an analysis by Lettenmaier et al. did not uncover significant associations for trend direction, except for population density. For phosphorus, 18% of the stations showed decreases, while 3% showed increases (table 2). The decreases occurred mainly in the Great Plains and in the East. Reasons for phosphorus trends also were unclear.

Phosphate concentrations do not reach levels of concern for drinking water; but a standard of 0.1 µg/l has been set to prevent eutrophication of estuaries (MacDonald et al. 1991). No standard has been set for freshwater, because risk of eutrophication is very

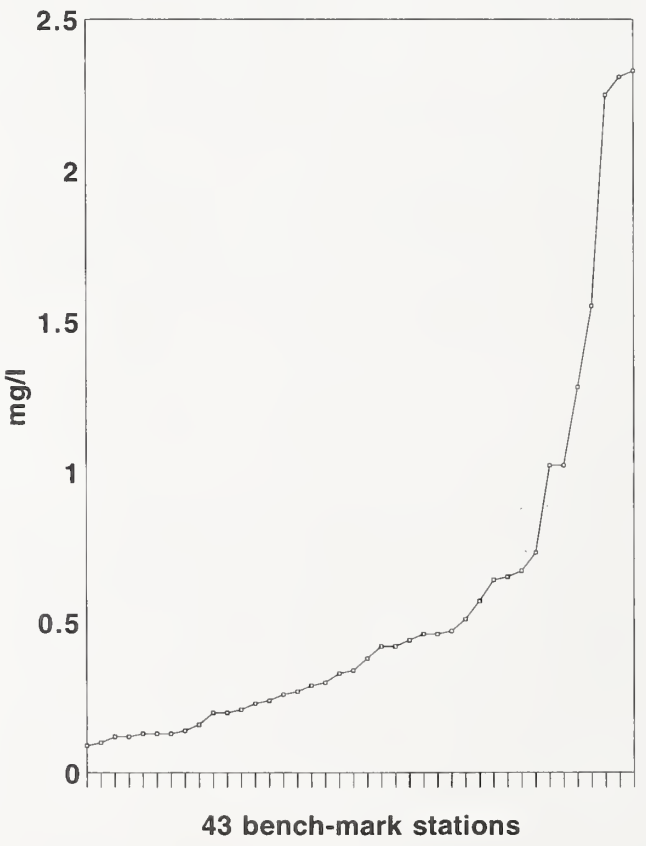

Figure 4.—Mean annual dissolved nitrogen concentration at 40 benchmark stations draining areas of forest and rangeland (stations shown in order of increasing dissolved nitrogen)

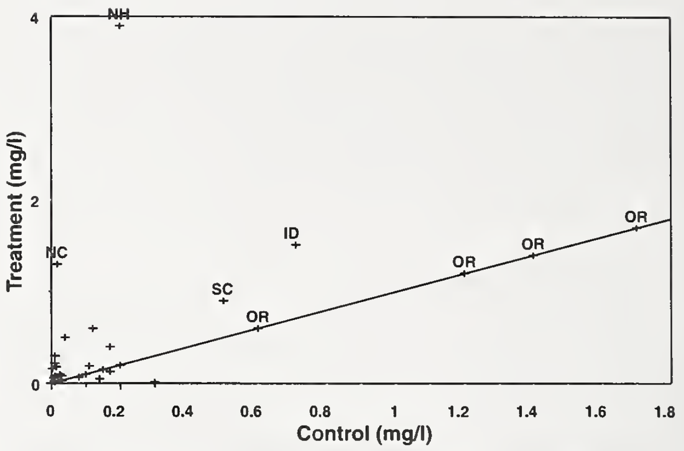

Figure 5.—Response of streamwater nitrate-N concentrations to forest harvesting at experimental watersheds (see table 4 for details) (state abbreviations identify watersheds with largest concentrations)

dependent on location; a level of 0.05 mg/l may be sufficient to protect lakes (MacDonald et al. 1991). Nitrate-N concentrations above 10 mg/l are unacceptable for drinking water because of risks to infants (EPA 1986). No toxicity standards for nitrate have been developed for protection of aquatic ecosystems; but achievement of the drinking water standard also should protect aquatic ecosystems (Bisson et al. 1992).

As shown in figure 3, nitrogen concentrations in streams draining forest areas are typically well below 1 mg/l. This is corroborated at the bench-mark and experimental watersheds. Only five of the bench-mark watersheds exceed this level (with mean levels ranging from 1.3 to 2.3 mg/l) (fig. 4); and they all drain largely rangeland areas, not forests. Among 30 experimental control watersheds that measured nitrogen concentrations, only 3 in Oregon catchments had nitrate levels exceeding 1 mg/l (fig. 5).

Effects of Management

The chemistry of water flowing through forests changes as water passes through the canopy, soil, and subsoil into streams. In the canopy, interception (and, therefore, evaporation) of precipitation concentrates dissolved elements. Forest harvesting reduces interception losses, allowing more water to reach the soil, thereby diluting nutrient concentrations, all else equal. Several processes alter the chemistry of water moving through mineral soils, including uptake of water and chemicals by tree roots, mycorrhizae, and microbes, release of chemicals from decomposition, and exchange reactions releasing and

Table 3.—Effect of harvest on phosphorus conecntration.[1]

State	Catchment	No. watersheds Cont.	Treat.	Treatment[2]	Concent.($\mu g/l$)[3] Cont.	Treat.	Diff.	Statistical significance[4]
OR	H.J. Andrews (6,7,8)	1	2	CC, burn	22	18	-4.0	
OR	H.J. Andrews (1,2)	1	1	CC	26	24	-2.0	
OR	Bull Run	4	10	CC	5.7	6.0		
0.3	yes for 1 of 10							
NH	Hubbard Brook	1	1	CC, herb.	1.18	1.56	0.4	
OR	High Ridge	1	3	CC, burn	15.5	16.8	1.3	yes for 1 of 3
ID	Horse Creek	5	5	various	9.7	19.4	9.7	
OR	Baker Creek	1	2	harvest	~50	~60	~10	
OR	H.J. Andrews (1,2)	1	1	burn5	16	39	23.0	
OR	Coyote Creek	1	3	various	~80	~40	~40	
GA	Piedmont	1	1	CC				no
ID	Twin Lakes	6	3	various				no
OR	Santiam	1	1	CC				no
OR	H.J. Andrews (2,10)	1	1	CC				no
OR	Alsea	1	2	CC				no
WA	Olympic peninsula	1	2	harvest				no
BC	Carnation Creek	1	1	harvest				no
BC	Haney	1	2	CC, burn	<20	<20		no

From Salminen and Beschta (1991), who summarized all entries. The first set of studies is ordered in terms of increasing difference between treatment and control concentrations. Means were not reported by Salminen and Beschta for studies in the second set.

[2]*In most cases, clearcuts (CC) were only over part of the watershed. In some cases, only "harvest" was listed by Salminen and Beschta.*

[3]*Mean annual concentration over various measurement periods, from 1 to over 10 years depending on the study. Where more than watershed is summarized, simple averages across watersheds are reported. In most cases, total phosphorus is listed; in some cases only orthophosphorus.*

[4]*The results of statistical tests of significance are listed where reported by Salminen and Beschta.*

[5]*This burn followed the clearcut by one year.*

absorbing chemicals. Forest floor disturbances resulting from burning and harvesting can allow precipitation to reach mineral horizons unaltered. Harvesting also reduces nutrient uptake, and also may increase soil decomposition rates, increasing the pool of available nutrients in the mineral soil.

Although forest practices may elevate the concentrations of many chemicals in streamwater, only the concentrations of phosphate and nitrate are of significant concern in forestry.

Phosphate. Salminen and Beschta (1991) comprehensively reviewed the effects of forest management on phosphate concentrations. Table 3 briefly summarizes the studies they reviewed on the effect of timber harvest on phosphate. Most studies found that mean concentrations were very similar for both treatment and control watersheds, and that differences were not statistically significant. Mean concentrations usually were well below the 0.05 mg/l level that has been suggested as sufficient to protect lakes from eutrophication. Salminen and Beschta concluded that changes in stream phosphate concentrations resulting from forest harvesting are uncommon, unless accompanied by high-intensity slashburns, and that even following such slashburns, the increases are short-lived.

Nitrate with harvest. Changes in annual streamwater nitrate with harvest are summarized in table 4, for 31 studies in the U.S. and southern Canada. Each row reports on a pair of watersheds, one control and one harvested. Where more than one level of harvesting intensity was examined, table 4 includes the most severe treatment. Where the period of record after harvesting covers more than one year, table 4 lists the maximum year. Annual concentrations do not show short-term pulses of higher concentrations that may be of chief concern. However, annual concentration was the most generally reported result, and allowed the broadest comparison among study sites. Figure 5 depicts the treatment and control levels for each case in table 4.

Table 4.—Effects of forest practices on annual streamwater nitrate concentrations (mg-N/l).[1]

State	Catchment	Concentration (mg/l) Control	Treat.	Diff.	Treatment	Reference
MN	Marcell	0.3	0.015	-0.29	80% clearcut	Verry (1972)
MT	Bitteroot	0.17	0.13	-0.04	100% clearcut	Bateridge (1974)
OR	HJ Andrews	0.08	0.07	-0.01	WS-10 100% clearcut	Sollins and McCorrison (1981)
SC	Santee	0.1	0.1	0.0	Prescribe burned	Richter et al. (1982)
WV	Fernow	0.2	0.2	0.0	100% clearcut	Aubertin and Patric (1974)
ID	Priest River	0.15	0.15	0.0	Clearcut unit beside stream	Snyder (1975)
OR	Alsea	1.2	1.2	0.0	88% clearcut	Brown et al. (1973)
OR	Coast range	0.6	0.6	0.0	Herbicide/clearcut	Miller and Newton (1983)
OR	Coast range	1.7	1.7	0.0	100% clearcut/herbicide	Miller and Newton (1983)
OR	Coast range	1.4	1.4	0.0	Herbicided	Miller and Newton (1983)
BC	Okanagan	0.03	0.03	0.0	16% clearcut	Hetherington (1976)
FL	Bradford	0.03	0.04	0.01	80% clearcut	Riekirk (1983)
ID	Silver Creek	0.01	0.018	0.01	23% clearcut	Clayton and Kennedy (1985)
UT	Chicken Creek	0.008	0.025	0.02	13% clearcut	Johnston (1984)
CO	Fraser	0.006	0.06	0.05	33% clearcut	Stottlemyer (1987)
PA	Leading Ridge	0.03	0.08	0.05	44% clearcut	Lynch et al. (1975)
OR	Bull Run	0.01	0.08	0.07	25% clearcut/burned	Fredriksen et al. (1975)
MT	Bitteroot	0.11	0.19	0.08	100% clearcut	Bateridge (1974)
OR	Coyote Creek	0.025	0.1	0.08	100% clearcut	Harr et al. (1979)
GA	Grant Forest	0.14	0.05	0.09	100% clearcut	Hewlett et al. (1984)
OR	High Ridge	0.003	0.162	0.16	100% clearcut	Tiedemann et al. (1988)
ID	Priest River	0.015	0.18	0.17	Clearcut unit beside stream	Snyder (1975)
AZ	Beaver Creek	0.01	0.22	0.21	100% clearcut	M. Ryan, pers. comm.
MT	Bitteroot	0.17	0.4	0.23	100% clearcut	Bateridge (1974)
TX	Cherokee Cty	0.01	0.3	0.29	100% clearcut	Blackburn et al. (1986)
SC	Georgetown	0.5	0.9	0.4	Drained	Askew and Williams (1986)
BC	UBC Research For.	0.04	0.5	0.46	100% clearcut	Feller and Kimmins (1984)
NB	Nashwaak	0.12	0.6	0.48	100% clearcut	Krause (1982)
ID	Priest River	0.71	1.51	0.8	Clearcut unit beside stream	Snyder (1975)
NC	Coweeta	0.018	1.3	1.28	WS-7 clearcut	Swank (1988)
NH	Hubbard Brook	0.2	3.9	3.7	100% clearcut	Hornbeck et al. (1987)

[1]*Adapted from Binkley and Brown (1993a). Cases ordered in terms of increasing difference between treatment and control concentrations. Where concentrations were reported for more than 1 year, the year of maximum treatment concentration is listed.*

The net effect of harvest on nitrate usually was to increase concentrations somewhat. However, about 70% of the studies found that annual concentrations of nitrate remained below 0.5 mg-N/l, for both control and harvested watersheds. Two patterns were apparent for the sites with greater concentrations. One group, primarily the forests of red alder and Douglas-fir in the Oregon Coast range, and the high elevation forests of red spruce and beech in the Appalachian Mountains, showed high levels of nitrate in control watersheds. No response to treatment was apparent in the Oregon Coast range forests (table 4). For Douglas-fir in the Pacific Northwest, enough studies have been done to show that the risks of nitrate pollution are small (Norris et al. 1991, MacDonald 1991). The evidence is less clear in the Appalachians; but the high baseline levels (which average about 5 mg/l at 1500m elevation, Silsbee and Larson 1992) suggest that any change in those forests, such as harvesting or further decline in vigor, could elevate nitrate concentrations close to allowable water quality standards. The other group, particularly the northern hardwood forests at Hubbard Brook, and Southwestern chaparral watersheds, showed substantial nitrate increases after some treatments, such as strip-cutting at Hubbard Brook (Hornbeck et al. 1987) and conversion of chaparral to grass (Riggan et al. 1985, Davis 1987). This Hubbard Brook example involves normal forest harvesting, **not** the inhibition of regrowth with herbicides, which produced even greater nitrate-N losses; (Likens et al. 1970). In no cases did the average annual nitrate concentration exceed the drinking water standard of 10 mg-N/l; however, the alder/conifer watersheds and the harvested Hubbard Brook watersheds had occasional pulses of nitrate that exceeded the standard.

Table 5.—Effects of forest fertilization on maximum streamwater nitate-N concentrations (peak observation for control and post-fertilization periods, mg nitrate-N/l).[1]

State	Catchment	Concentration (mg/l) Control	Treat.	Diff.	Treatment	Reference
OR	Siuslaw Nat. Forest	0.16	0.13	-.03	225 kg-N/ha as urea	EPA (1980)
OR	Willamette Nat. Forest	0.1	0.13	0.03	225 kg-N/ha as urea	Fredriksen et al. (1975)
WA	Olympic Nat. Forest	0.005	0.04	0.035	225 kg-N/ha as urea	Fredriksen et al. (1975)[2]
WA	Entiat Exp. Forest	<0.1	0.15	<0.05	110 kg-N/ha as urea[3]	Tiedemann et al. (1978)
WA	Olympic Nat. Forest	0.03	0.12	0.09	225 kg-N/ha as urea	Fredriksen et al. (1975)[2]
OR	Coyote Creek	0.005	0.18	0.175	225 kg-N/ha as urea	Fredriksen et al. (1975)
OR	Yamhill River	0.1	0.4	0.3	225 kg-N/ha as urea	Fredriksen et al. (1975)
WA	Olympic Nat. Forest[4]	<0.01	0.07-0.72	<0.71	225 kg-N/ha as urea	EPA (1980)
OR	Siuslaw Nat. Forest	0.5	2.1	1.6	225 kg-N/ha as urea	Fredriksen et al. (1975)
AK	Falls Creek	0.24	1.7	1.46	210 kg-N/ha as urea	Meehan et al. (1975)
WA	Various locations	<0.3	2.7	<2.4	65 to 225 kg-N/ha as urea	Bisson (1982)
AK	Three Lakes	0.20	2.7	2.5	210 kg-N/ha as urea	Meehan et al. (1975)
WA	Louse Creek	0.1	3.0	2.9	200 kg-N/ha as urea	Bisson et al. (1992)
OR	Siuslaw Nat. Forest	4.3	7.6	3.3	225 kg-N/ha as urea	EPA (1980)
WA	Ludwig Creek	1.0	6.0	5.0	200 kg-N/ha as urea	Bisson et al. (1992)
BC	Lens Creek	0.05	9.5	9.45	225 kg-N/ha as urea	Hetherington (1985)
WV	Fernow Exp. Forest	0.2	>10 for 3 wks	>9.8	340 kg-N/ha as ammon. nitrate	Helvey et al. (1989)[5]
WV	Fernow Exp. Forest	0.2	16	15.8	225 kg-N/ha as urea after cc	Kochenderfer and Aubertin (1975)
BC	Mohun Lake	—	0.16		N fertilization with 50m buffer	Bisson et al. (1992)
BC	Mohun Lake	—	0.39		N fertilization with no buffer	Bisson et al. (1992)

[1]*Peak nitrate concentrations, which usually occurred within two months of the fertilizer application. Adapted from Binkley and Brown (1993a). Listed in order of increasing difference between treatment and control concentration.*

[2]*Also EPA (1980).*

[3]*Plus ammonium sulfate.*

[4]*13 studies.*

[5]*Also Edwards et al. (1991).*

Martin et al. (1984) summarized the effects of conventional clearcutting practices on water quality, in 38 watersheds from around New England. Watersheds were not paired as controls and treatments; so, these data were not included in figure 5. Streamwater concentrations of nitrate for unharvested watersheds were near 0 mg-N/l for central hardwood forests, between 0.15 and 0.5 mg/l for the conifer forests, and 0.15 to 1.0 mg/l for northern hardwoods. Clearcutting from 20% to 100% of the watershed resulted in no prolonged nitrate increases in streamwater draining central hardwood or conifer forests. Streamwater nitrate-N concentrations for northern hardwood forests showed no effect of clearcutting, if <70% of the watershed was harvested, and either no change or an increase up to 2.0 mg/l in completely harvested watersheds. The authors found no cases where the increase in nitrate concentrations were as marked as those found after harvesting northern hardwoods forests, at Hubbard Brook, in the White Mountains of New Hampshire.

To summarize, the evidence indicates that timber harvesting does not degrade water quality by increasing nitrate concentrations in streamwater, with the possible exception of the Hubbard Brook forests. Even in the Hubbard Brook cases, rapid revegetation cut the high nitrate losses. Also, downstream dilution reduces nitrate concentrations, usually to within safe levels, before water supply intakes are reached. However, high rates of atmospheric deposition of nitrogen compounds already may be generating high nitrate-N losses from some high-elevation forests in the eastern U.S., such as the Smoky Mountains (Silsbee and Larson 1980, Johnson and Lindberg 1992). Current generalizations about low-nitrate losses from eastern forests may not apply within several decades (cf. Aber et al. 1990, Johnson 1991).

Nitrate with fertilization. Fertilization adds large pulses of nutrients that may exceed the immediate uptake ability of trees. Table 5 summarizes peak nitrate concentrations for 14 forest fertilization studies, mostly from the Pacific Northwest. Peak concentrations usually occurred within 2 months of fertilizer applications. The effects of fertilization varied substantially among studies. At sites in Oregon, Washington, and Alaska,

peak nitrate-N concentrations after fertilizer application ranged from 0.1 to 2.7 mg/l. These were short-duration peaks, below levels of serious concern. The patterns were similar from another 28 studies in the Northwest, which showed peak nitrate-N concentrations of about 4 mg/l (EPA 1980). Fredriksen et al. (1975) concluded that fertilization of such areas does not raise nitrate concentrations to toxic levels, and poses no threat to stream water quality.

Hetherington (1985) examined streamwater nitrogen concentrations after fertilization with urea, on Vancouver Island, British Columbia; no effort was made to minimize application over streams within the unit. Nitrate concentrations peaked about 2 months after application, as heavy rains drained the soils. The maximum nitrate-N concentration reached 9.5 mg/l. Nitrate concentrations returned to pretreatment levels during the first winter, with a minor peak occurring 1 year after fertilization.

The effects of forest fertilization on streamwater concentrations of nitrate have been examined for only one location outside the Pacific Northwest. At the Fernow Experimental Forest in West Virginia, Kochenderfer and Aubertin (1975) reported peak nitrate-N concentrations of 16 mg/l in October, following fertilization with 225 kg-N/ha as urea, after clearcutting Watershed 1. Helvey et al. (1989) and Edwards et al. (1991) reported that fertilization of an intact forest with N (340 kg-N/ha as ammonium nitrate) raised streamwater nitrate-N concentrations above the 10 mg/l drinking water standard for 3 weeks, during the autumn after application.

In general, others have found that careful fertilization does not increase streamwater concentrations of nitrate-nitrogen to potentially toxic levels (Fredriksen et al. 1975, Miller and Fight 1979, Hetherington 1985, Norris et al. 1991). Careful fertilization avoids excessive application rates and times applications so they do not coincide with heavy runoff. Forest fertilization in the Pacific Northwest and British Columbia is unlikely to degrade water quality below drinking water standards. The situation is less clear for forests in the East; the one site that has been examined showed excessive nitrate concentrations in two separate trials.

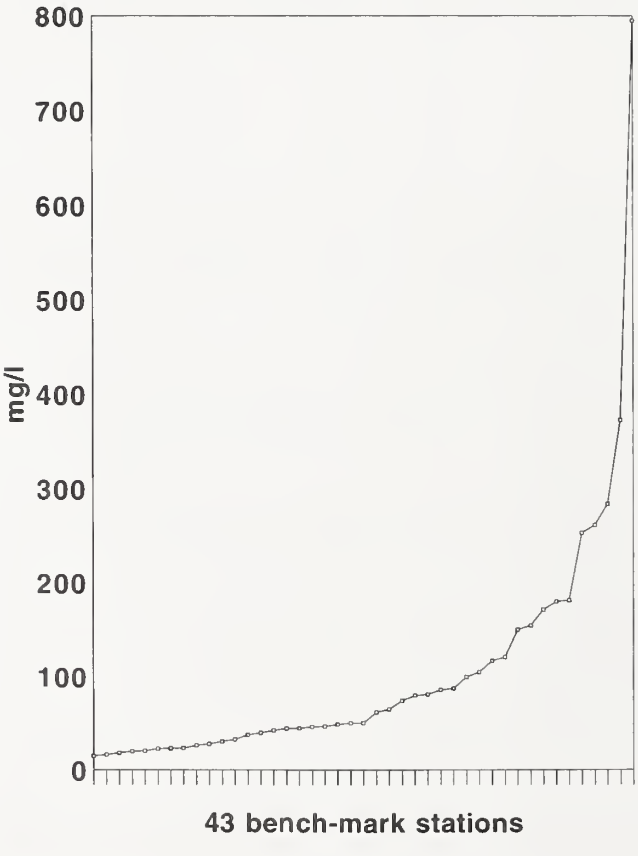

Figure 6.—Mean annual dissolved solids concentration at 43 benchmark stations draining areas of forest and rangeland (stations shown in order of increasing dissolves solids)

Dissolved Solids

Background

Salts enter the Nation's waterways from natural dissolution of rock and soil, from atmospheric precipitation (which contains ions from both natural and human sources), and from activities such as municipal and industrial water treatment releases, de-icing roads, and the concentrating effects of irrigation. Peters (1984) reported, based on analysis of the USGS's nationwide NASQAN data, that the primary determinants of dissolved solids concentrations are rock type, precipitation quantity, precipitation quality, and, to a lesser extent, human population density. Hem (1989) emphasized the importance of rock type and precipitation quality, as well as human causes, such as salting roads and irrigation, in determining concentration of dissolved solids. Among the bench-mark watersheds, mean annual total dissolved solids concentrations are mostly below 200 mg/l (fig. 6). The 5 stations that exceed 200 mg/l, drained watersheds containing a mixture of forest and range vegetation types, and were located throughout the U.S.

Smith et al. (1987) and Lettenmaier et al. (1991) found many more increases than decreases in dissolved solids (table 2). Increases were most common in the eastern half of the country. Smith et al. suggested the changes were affected by human waste discharges, salt use on roads, and surface coal production.

Effects of Management

Forest and range vegetation apparently do not add significantly to the salt content of the Nation's rivers. The effects of forest management on concentrations of dissolved solids are so slight, in relation to the levels of concern for water uses, that this area has received little study. The principal effect of forest management may be the effect that vegetation density has on concentration of dissolved solids that reach the water from other sources. For example, Brown et al. (1990) estimated the beneficial effect of increases in runoff from timber harvesting, on the dilution of salts, in the Lower Colorado River Basin, where salt content is a significant concern.

Sediments

Background

Based on the 1982 National Resource Inventory (SCS 1984) estimates of erosion at nearly 800,000 points on nonfederal rural lands, and on sediment transport and delivery predictions, discharge rates into rivers and streams from cropland were estimated to be more than five times the rate from forest land (Gianessi et al. 1986) (fig. 7). The rangeland rate was 1.5 times the forest rate. Of the total discharge from non-federal rural lands, 57% was estimated to originate on cropland, 16% on rangeland, 10% on forest land, 5% on pastures, and 12% on other land (mines, quarries, farmsteads, and other uses). Forests and range lands together, which occupy 2/3 of the Country, were estimated to contribute one-fourth of the sediments reaching the Nation's waters from private land. However, forests also tend to occupy the steepest portions of landscapes, suggesting that portions of forested drainage basins may be especially susceptible to erosion.

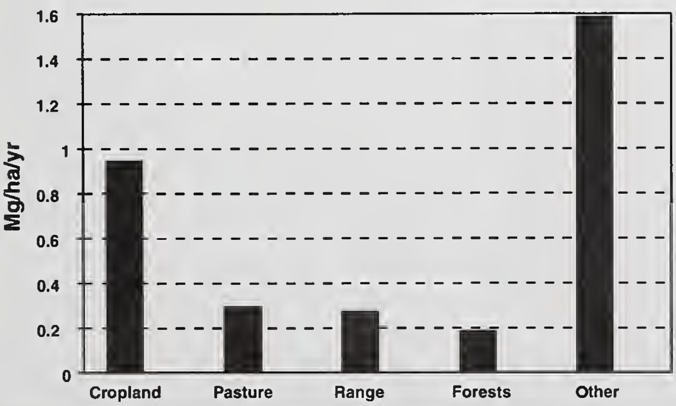

Figure 7.—Annual sediment yields from non-federal rural lands into U.S. rivers (from Gianessi et al. 1986) ("Other" includes mines, quarries, farmsteads, and other intensively used sites.)

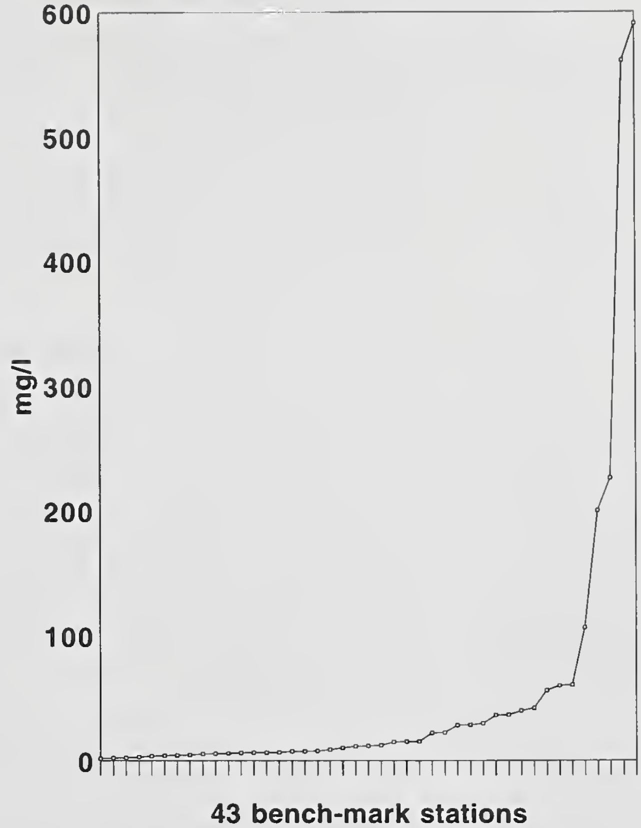

Figure 8.—Mean annual suspended sediment concentration at 43 benchmark stations draining areas of forest and rangeland (stations shown in order of increasing suspended solids)

Lettenmaier et al. (1991) found few significant trends in suspended sediments; 12% of the stations showed decreases, and 8% showed increases (table 2). No associations of trend with land cover were found. Smith et al. (1987) reported that 14% of the stations decreased and 16% increased. Most of those increases occurred in the Columbia and Mississippi basins. Trends in suspended sediment concentrations were not significantly associated with total basin soil ero-

Table 6.—Effect of harvest and road construction on suspended sediment concentration.[1]

State	Catchment	Concent. (mg/l) control	treat.	diff.	Measure[2]	Treatment[3]	Reference
TX	Cherokee Co.	79	36	~43	Max. annual avg. in 4 years	100% clearcut, chopped	Blackburn et al. (1986)
OR	Santiam River[4]	71	50	~21	9-year average	CC patches in large area	Sullivan (1985)
OR	Bull Run	2.4	2.4	0	Max. annual avg. in 8 years	Harvest 25%	Fredriksen et al. (1975)[5]
OR	Bull Run	2.4	2.4	0	Max. annual avg. in 10 yrs	Harvest 25%, burn	Fredriksen et al. (1975)[5]
OR	Coyote Creek	~20	~20	~0	??	Shelterwood cut	Harr et al. (1979)[6]
OR	Coyote Creek	~20	~20	~0	??	Patchcuts	Harr et al. (1979)[6]
FL	Bradford Co.	3	4	1	2-year average	~80% clearcut, min impact	Riekirk (1983)
ID	Priest River[4]	4.5	6.4	2	1-year average	Harvest	Snyder et al. (1975)
PA	Leading Ridge	1.7	5.9	4	3-year average	Harvest 43%	Lynch et al. (1975)[7]
OR	Alsea	1	6	5	5-year average	25% clearcut	Fredriksen et al. (1975)[8]
OR	Alsea	1	10	9	5-year average	100% clearcut	Fredriksen et al. (1975)[8]
FL	Bradford Co.	3	13	10	2-year average	~80% clearcut, max impact	Riekirk (1983)
ID	Priest River[4]	3	16	13	1-year average	Harvest	Snyder et al. (1975)
ID	Priest River[4]	7.1	37	30	1-year average	Harvest	Snyder et al. (1975)
OR	Alsea	95	136	41	7-year average	road const, 25% CC, burn	Beschta (1978)
OR	Alsea	95	146	51	7-year average	road const, 82% CC, burn	Beschta (1978)
SC	Clemson E.F.	19	72	53	max. annual avg. in 3 yrs	Burn, 100% clearcut	Van Lear et al. (1985)
PA	Leading Ridge	1.7	80	78	2-year average	Clearcut, herbicide	Lynch et al. (1975)[7]
AK	Southeast	35	117	82	2-year average	Harvest, burn	Stednick et al. (1982)
TN	Upper Coast Pl.	82	183	101	Max. stormflow in 3 years	100% clearcut	McClurkin et al. (1985)
OR	Coyote Creek	~20	170	~150	??	Clearcut	Harr et al. (1979)[6]
WA	Hansel Creek[9]	3.7	178	174	Max. annual avg. in 3 years	Harvest, road construction	Fowler et al. (1988)
OR	HJ Andrews	9	240	231	6-year average	100% clearcut, burn	Fredriksen et al. (1975)
MS	Upper Coast Pl.	2127	2471	344	Max. annual avg. in 2 years	100% clearcut, chopped	Beasley (1979)
OR	HJ Andrews	9	430	421	6-year average	25% clearcut, road const.	Fredriksen et al. (1975)
MS	Upper Coast Pl.	2127	2808	681	Max. annual avg. in 2 years	100% clearcut, sheared	Beasley (1979)
MS	Upper Coast Pl.	2127	2837	710	Max. annual avg. in 2 years	100% clearcut, sheared	Beasley (1979)
TX	Cherokee Co.	112	1158	1046	Max. annual avg. in 4 years	100% clearcut, sheared	Blackburn et al. (1986)

[1]Adapted from Binkley and Brown (1993a). Cases are listed in order of increasing difference between treatment and control concentration. With one exception (the Upper Coastal Plain site in Tennessee), concentrations are for annual periods.

[2]The years refered to are the initial post-treatment years. When "Max. annual avg." is reported for some time period, the concentrations reported are for the year with the maximum concentration from the treatment watershed.

[3]Clearcut = CC.

[4]A separate control watershed was not available. Control refers to upstream of the treatment area, and treatment to within or downstream of the treatment area.

[5]Also Fredriksen and Harr (1988).

[6]Also Adams and Stack (1989).

[7]Also Lynch and Corbett (1990).

[8]Also Brown et al. (1973).

[9]A separate control watershed was not available. Control refers to prior to treatment, and treatment refers to after treatment.

sion rates, but increases in concentration were significantly related to the fraction of total soil erosion contributed by cropland in the basin and to the absolute magnitude of cropland erosion in the basin. Trends were not statistically associated with erosion rates on forests or rangeland, although the increases in the Columbia River Basin occurred mainly in areas with significant forest cover and timber harvest.

Suspended sediments are measured as the weight of particles retained on a filter paper after the water has been filtered (typically in units of mg of sediment per l of water), or as turbidity, a measure reported in terms of the amount of light scattered by a water sample. Water quality standards may include suspended sediment criteria, such as 500 mg/l for Oregon (Moore et al. 1979).

As figure 8 indicates, mean annual suspended sediment concentrations in largely undisturbed streams draining forest and range lands are below 100 mg/l in most areas. Further, the higher average levels among the bench-mark watersheds were not found in forested areas; all 5 of the stations with

concentration levels above 100 mg/l drain largely rangeland watersheds.

Table 6 lists the concentrations of suspended sediments for 19 control experimental watersheds across the U.S. (fig. 9). In all but one case, the concentrations in table 6 are annual figures. The most common pattern across the U.S. is for very low sediment concentrations in streams draining undisturbed forested watersheds. Annual average concentrations are typically less than 5 mg/l, with stormflow peaks of up to 100 mg/l. This common pattern does not apply to all regions, however. Substantial areas of the Piedmont and Upper Coastal Plain of the Southeastern U.S. typically show much higher concentrations of suspended sediments. For example, an undisturbed forest of hardwoods and pine in Mississippi showed average annual concentrations of suspended sediment of 393 mg/l in one year, and more than 2000 mg/l in another (Beasley 1979). In Cherokee County, Texas, Blackburn et al. (1986) found that sediment concentrations ranged from 31 to 213 mg/l (averaging 112 mg/l), in an undisturbed pine forest. The intermittent flow of some of these streams contributed to the high concentrations. Some larger streams and rivers in other regions also may show relatively high concentrations of suspended sediment. For example, Sullivan (1985) found that the Middle Fork of the Santiam River, in the Oregon Cascades, averaged 71 mg/l over 9 years.

Effects of Management

Sediment enters streams in three basic ways: (1) detachment of soil particles by the impact of raindrops or water flowing across the soil surface, (2) mass movement of soil on steep slopes, e.g., debris slides or avalanches (Swanston 1991), and (3) stream channel bank erosion. Each of these types of processes of course occurs naturally, and can be aggravated by human forces. First, removal of vegetative

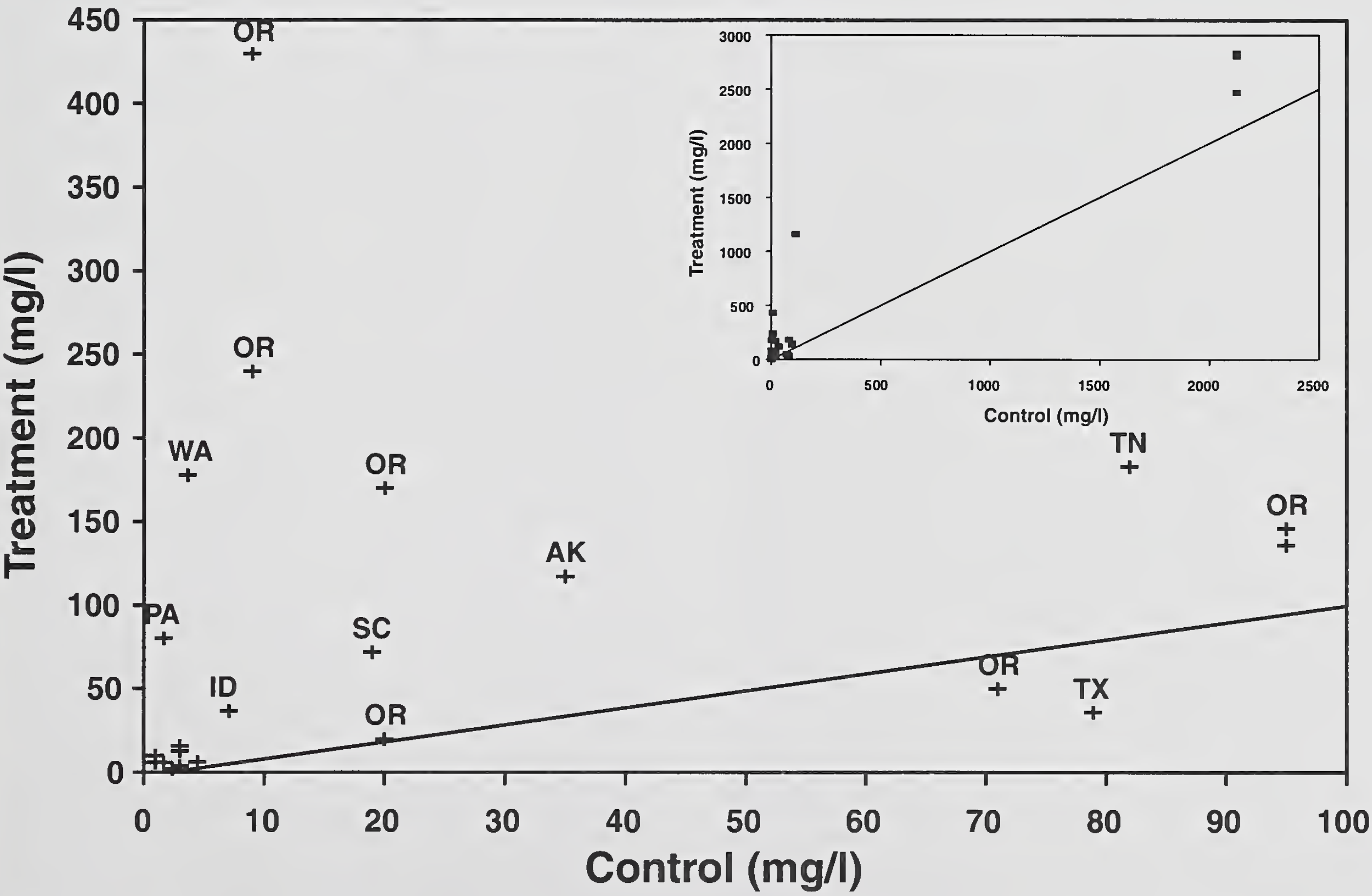

Figure 9.—Response of suspended sediment concentrations to forest harvesting at experimental watersheds (see table 6 for details) (from Binkley and Brown 1993a)

cover (e.g., by road building, harvest, grazing, mechanical disturbance, or fire) increases the impact of raindrops and reduces trapping of mobilized sediments. Some activities (e.g., harvest equipment or concentrations of cattle) compact the soil, reducing infiltration and increasing the chances of overland flow. Second, harvesting and road construction can increase the likelihood of mass soil movements in several ways. Harvesting typically leads to greater soil moisture because of reduced interception and transpiration; and wetter soils are less stable than drier soils. The decay of tree roots after harvesting allows more slope failures. Road construction collects and concentrates water moving downslope, and increases slope angles on both the cut and fill sides (Swanston and Swanson 1976). In the H.J. Andrews Experimental Forest, Swanson and Dyrness (1975) found that forested slopes accounted for about 35 m^3 of debris flow material per km^2 annually, compared with 130 m^3/km^2 for clearcut slopes, and 1800 m^3/km^2 for roaded areas. Finally, stream channel banks can be damaged by harvesting and road construction equipment, as well as by cattle when they congregate around water sources.

Concentrations of suspended sediments often increase after management activities, such as road construction, harvest, and site preparation. The increases can vary greatly, including some situations where no increases occur (table 6). The cases, in table 6, with low harvest impacts tend to be those where efforts were made (e.g., through use of best management practices) to control erosion. For example, in the Leading Ridge watershed in Pennsylvania (Lynch et al. 1975, Lynch and Corbett 1990), the harvesting operation did not prevent an increase in sediment concentrations, but the increase was kept to a minimum (1.7 mg/l for the control watershed, 5.9 mg/l for the harvested watershed) by following best management practices (BMPs). The road construction, harvesting, and slash-burning in Fox Creek, in the Bull Run Watershed in Oregon, did not increase sediment concentrations beyond the 0.6-2.4 mg/l level of the control watershed (Fredriksen et al. 1975, Fredriksen and Harr 1979).

In other cases, and without imposition of BMPs, substantial (and varying) increases in sediment concentration may occur. For example, harvesting the hardwood/pine forest in Mississippi was followed by a variety of site preparation treatments (Beasley 1979). In the first post-treatment year, the concentrations of suspended sediment rose from 2127 mg/l in the control to more than 2800 mg/l for the more intensive site preparation treatments. In the second year, the sediment concentration declined to 393 mg/l in the control, but remained more than 2300 mg/l for the most intensive treatment; concentrations after intermediate intensity treatments ranged between 600 and 900 mg/l. The causes of the high sediment concentrations include the intermittent nature of these small streams (flow stops for substantial periods between storms), the effect of forest removal on streamflow (increased volume and period of flow), and an apparent failure to retain a vegetated buffer strip next to the streams.

The major conclusion from these watershed experiments is that suspended sediment concentrations vary greatly. As seen in figure 9, the baseline suspended sediment concentrations vary greatly across sites, as do the impacts of treatment. Differences in soil type, slope, and weather influence the variation among the control watersheds. Treatment effects do not appear to be related to baseline concentrations, and, in general, are most likely to be related to the severity of the treatment (especially to road and skid trail activity), and to case-specific storms after the treatments.

The most important ecological impacts of forest practices on sediment-related features involve physical changes in stream structure or condition (MacDonald et al. 1991). These changes include increased content of fine particles in gravel beds, erosion of stream banks, increases in stream width, decreases in stream depth, and fewer deep pools. These physical features of stream structure may provide a better focus for monitoring and assessing forest practice impacts than direct monitoring of sediment concentrations in the water column (MacDonald et al. 1991).

Poor management of livestock grazing greatly accelerates erosion. In the Wasatch Range in Utah, overgrazing by sheep increased soil erosion from about 0.1 mg/ha to more than 15 mg/ha (Noble 1963, Brown 1989). Although many studies have characterized grazing impacts on vegetation, soil physical properties, and small-scale erosion, few studies have directly examined the connections between grazing impacts and water quality.

Perhaps the major conclusions that can be drawn from the variety of studies in table 6 and the other studies mentioned here are that (1) concentrations of

suspended sediment differ substantially throughout the U.S.; (2) the response to forest practices depends greatly on the particular situation and the details of the treatments that are applied; (3) use of BMPs substantially reduces negative impacts of management; (4) mass failures and erosion of stream banks may be particularly significant and are not always avoidable; and (5) depending on soil and slope conditions, extreme precipitation events will degrade water quality for both managed and unmanaged areas. The failure of an undercut bank, or degradation of streamside vegetation by grazing animals, may have much larger impacts on stream sedimentation than activities dispersed throughout a watershed.

Table 7.—Annual use of pesticides in agriculture and forestry in 1980.

	Total (mg/yr)[1]		Application rate ($kg/km^2/yr$)	
	Agriculture	Forests[2]	Agriculture[3]	Forests[2,4]
Herbicides	202,030	169	131.4	0.220
Fungicides	22,700	9	14.8	0.011
Insecticides	138,924	71	90.3	0.092

[1]*For 1980, as presented by Norris et al. (1991, table 1.7).*
[2]*National forest land only.*
[3]*Land area used for crops taken from ERS (1989).*
[4]*Land area taken from U.S. Forest Service (1983).*

Toxics

Background

Pesticides, including herbicides, are among most common toxics used in rural areas. Agricultural uses dominate, but pesticides also are used on forests and rangelands. Table 7 gives a rough indication of the relative application rates, in 1980, on agricultural and national forest lands in the U.S. Agricultural use per land unit was roughly 1,000 times forest use for insecticides, 600 times for herbicides, and 1,300 times for fungicides. The short crop cycle in intensive farming, compared to silviculture, contributes to the much heavier use of such chemicals in agriculture. Most forest lands in the U.S. are not treated with pesticides during a typical crop cycle of 20-100+ years; and lands that are treated seldom receive more than one application per cycle. Furthermore, because erosion rates are generally lower on forested land, a smaller proportion of the chemicals that attach to soil particles are transported to streams. Regarding the comparison in table 7, only about 20% of the forests in the U.S. are in national forests; use rates on other forest land may be higher than on national forests. Also, the types of pesticides used on forest lands has been changing over the past 20 years towards less toxic chemicals (Norris et al. 1991).

Neither Smith et al. (1987) nor Lettenmaier et al. (1991) examined trends for pesticides in stream water. Lettenmaier et al. (1991) did examine trends for several trace metals, finding that decreases were much more common than increases, and that the decreases were spread across the U.S.(see table 2 for examples).

Effects of Management

Three aspects of pesticide application to forests may influence water quality: concentrations of pesticides in streams, response of stream chemistry to pesticide treatment, and effect of treatment on erosion. Pesticides may enter streams directly from the application, or by movement from the soil (either on soil particles, or dissolved in water).

Careful studies of insecticide concentrations, after applications for pests such as spruce budworm, have not been attempted. Some information is available on the effects of herbicide treatments on the concentrations of elements in streamwater. Fredriksen et al. (1975) summarized a range of studies with several herbicides, and concluded that concentrations in streams were too low to warrant concern (peaking at about 0.01 mg/l within hours of application, declining to < 0.001 mg/l after weeks). No reports have appeared of injury to stream biota from herbicide applications that followed regulatory guidelines (Newton and Norgren 1977, Norris et al. 1991).

Concentrations of herbicides in streams following forest application are generally less than 0.1 mg/l, and levels of >2 mg/l would be needed to affect stream flora. Use of herbicides to alter riparian vegetation could have a variety of indirect effects on streams, including increased light, decreased bank stability and altered inputs of organic matter. Little information is available on the combined indirect effects; but they are likely to be within the normal variations found with the development of watershed vegetation after natural disturbances.

Although the few studies that have measured the effects of forest pesticide applications on water quality indicate that, when proper precautions are followed, the impacts appear to be minor, the lack of studies suggests that caution is needed, along with further field measurement.

Water Temperature

Background

Most aquatic organisms have optimal temperature ranges; forest practices that change temperatures more than about 2°C from natural temperatures may alter development and success of fish populations (Hornbeck et al. 1984). Higher temperature in late winter and spring may accelerate progression among life history stages of fish and other aquatic organisms, whereas high temperatures combined with low flows in late summer could be detrimental to fish populations. Some fish species require a narrow range of temperature; others tolerate wider ranges.

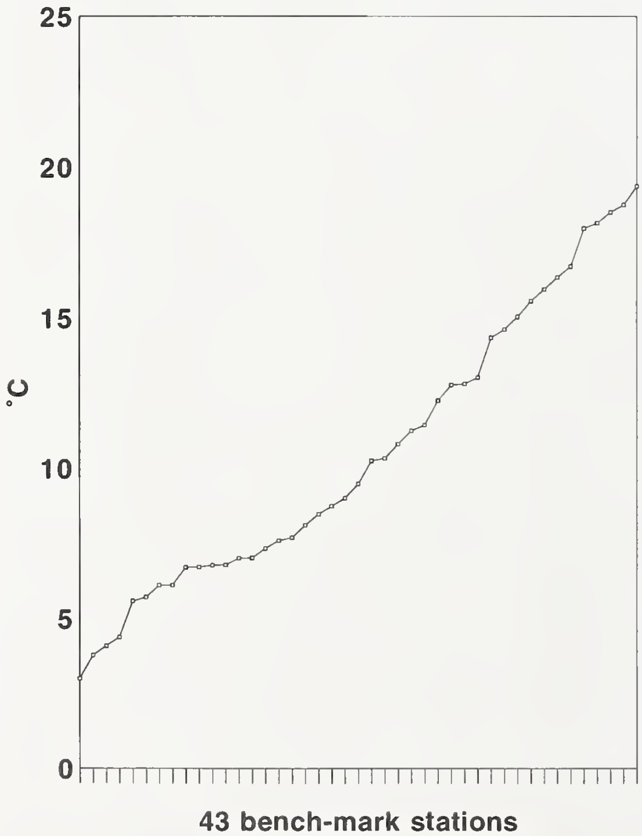

Figure 10.—Mean annual water temperature at 43 benchmark stations draining areas of forest and rangeland (stations shown in order of increasing temperature)

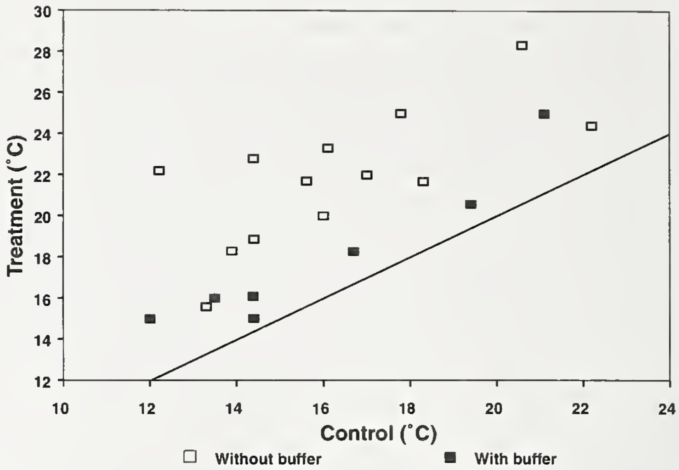

Figure 11.—Response of temperature maximums to harvest on experimental forest watersheds (see table 8 for details) (from Binkley and Brown 1993a)

Mean annual water temperature at the bench-mark stations ranges from 3° to 19°C (fig. 10), and largely reflects the surrounding air temperature. For the smaller experimental watersheds, table 8 reports temperature maximums. For the 18 unique control watersheds listed in table 8, temperature maximums range from 12° to 22°C. At these smaller sites, maximum temperatures are more sensitive to the presence or absence of shading vegetation.

Effects of Management

The effects of harvesting on stream water temperatures have been studied at many experimental watersheds. Several of the studies reported only the temperature change, not the control and treatment temperatures. Table 8 lists 20 comparisons for which both treatment and control temperature maximums were reported. The treatments differ considerably among these 20 sites, from complete overstory removal to clearcuts of only 25% of the watershed, and from leaving no trees along the stream (no buffer) to leaving a substantial buffer. Temperature maximums are reported for several different time periods, from single days during a summer month, to average daily for the hottest month, to average weekly for the entire growing season. Removal of forest canopies over streams increases radiation inputs and can raise maximum stream temperatures by 5° C or more.

Table 8.—Effect of harvest on temperature maximums.[1]

State	Temperature (°C)			Measure	Overstory treatment[2]	Reference
	control	treat.	diff.			
OR	14.4	15.0	0.6	A day in July	CC, with buffer	Brown et al. (1971)
PA	19.4	20.6	1.2	Avg. daily for hottest month	44% CC, with buffer (30 m)	Rishel et al. (1982)
OR	16.7	18.3	1.6	A day in July	CC, thin buffer	Brown et al. (1971)
WV	14.4	16.1	1.7	Avg. weekly for growing season	95% CC, thin buffer	Aubertin and Patric (1974)
OR	22.2	24.4	2.2	A day in July	CC, no buffer, reveg	Brown et al. (1971)
OR	13.3	15.6	2.3	A day in July	CC, no buffer	Brown et al. (1971)
OR	13.5	16.0	2.5	Avg. daily for hottest 3 weeks	25% CC, thin buffer	Harr and Fredriksen (1988)
OR	12.0	15.0	3.0	Avg. daily for hottest 3 weeks	25% CC, thin buffer	Harr and Fredriksen (1988)
NC	18.3	21.7	3.4	Avg. daily for hottest month	CC, no buffer	Swift and Messer (1971)
GA	21.1	25.0	3.9	Avg. daily for hottest month	CC, with buffer (12 m)	Hewlett and Fortson (1982)
NH	16.0	20.0	4.0	Avg. daily for hottest month	100% CC, no buffer	Likens et al. (1970)
OR	13.9	18.3	4.4	A day in July	CC, no buffer	Brown et al. (1971)
OR	14.4	18.9	4.5	A day in July	CC, no buffer	Brown et al. (1971)
BC	17.0	22.0	5.0	Avg. daily for July	60% CC, no buffer	Feller (1981)
OR	15.6	21.7	6.1	Avg. daily for hottest month	100% CC, no buffer	Levno and Rothacher (1967)
PA	17.8	25.0	7.2	Avg. daily for hottest month	85% CC, no buffer	Rishel et al. (1982)
OR	16.1	23.3	7.2	A day in July	CC, no buffer	Brown et al. (1971)
OR	20.6	28.3	7.7	A day in July	CC, no buffer	Brown et al. (1971)
OR	14.4	22.8	8.4	Avg. daily for hottest month	100% CC, no buffer	Levno and Rothacher (1969)
OR	12.2	22.2	10.0	Avg. daily for hottest month	100% CC, no buffer	Brown and Krygier (1969)

[1] *Taken from Binkley and Brown (1993a).*

[2] *All overstory treatments were clearcuts of some or all of the watershed. Where the portion of the watershed that was clearcut was not provided, we listed the treatment simply as clearcut (CC).*

The studies are listed in table 8 in order of increasing difference in maximum temperature. As seen in figure 11, which plots the key results of the table 8 studies, there is no apparent relationship between temperature increase and control watershed water temperature. A relationship between the temperature increase and vegetation treatment also is not apparent, with the exception that leaving a buffer to shade the stream appears to have a significant effect (fig. 11). Among the 10 sites (out of 20) with the lowest temperature increase, 7 left a buffer, and one was aided by quick revegetation along the stream.

The data in table 8 do not include some of the factors that affected measured temperatures, such as the presence, in some streams, of temperature changes from groundwater inputs, and the variations in buffer widths. Also, the table does not always indicate the maximum instantaneous temperature changes, because most published reports give average temperature maximums over a series of days. Nevertheless, the influence of buffer strips, and the general lack of other apparent influences, on the extent of the temperature increase after harvesting, demonstrate that retention of buffer strips along streams generally minimizes changes in temperature, in most cases below 2°C.

Further evidence of the effect of buffer strips is found in studies that compared treatments with and without buffer strips. At the Fernow Experimental Forest, in north-central West Virginia, clearcutting the hardwood forests but retaining a buffer resulted in no change in stream temperature; however, removal of the buffer 3 years later raised the mean temperature by about 2°C during the growing season (Patric 1980). In the Alsea watersheds, in the Coast Range of Oregon, retention of buffer strips of red alder and Douglas-fir along streams prevented any change in stream temperatures; but a lack of buffer strips allowed the monthly average water temperature to increase by 8°C, with a maximum of 16°C (Brown and Krygier 1970). However, commonly applied riparian leave strips are not always sufficient to avoid all harm. A detailed study in Washington concluded that riparian zone harvest restrictions do not always prevent increases in stream temperature that adversely affect fish (Sullivan et al. 1990). They suggested that leave strips should be maintained at minimum overstory densities, rather than at mini-

mum percentages of preharvest densities. See Welsch (1992) for more on the beneficial effects of riparian forest buffers.

Temperature increases are not always a problem. For example, at Carnation Creek, on the western side of Vancouver Island (Hartman et al. 1987), water temperatures increased after logging by about 4°C in summer, and 1°C in winter (Holtby 1988a,b). In this case, increased stream temperatures may have been beneficial to salmon populations, allowing earlier fry emergence and smolt migration.

Water Quality Protection Programs

The Federal Water Pollution Control Act of 1972 was the first federal law to focus significant attention on nonpoint source pollution. Section 208 of the Act required states to adopt an "area wide waste treatment management planning process" that was applicable to "all wastes generated within the area," including "silviculturally related nonpoint sources of pollution." The Clean Water Act of 1977 amended the 1972 Act by, among other things, authorizing a program of grants to help cover the costs to rural land owners of implementing what were called "best management practices" (33 U.S.C. 1288) to control nonpoint source pollution.

EPA regulations define best management practices (BMPs) as: "those methods, measures, or practices to prevent or reduce water pollution and include but are not limited to structural and nonstructural controls, and operation and maintenance procedures. BMPs can be applied before, during, and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters. Economic, institutional, and technical factors shall be considered in developing BMPs" (40 C.F.R. 35.1521-(4)(c)(1), 1984). Lynch et al. (1985) list examples of silviculture BMPs, and Chaney et al. (1990) list examples of grazing BMPs. Not all states use the term "best management practices" (BMPs); for example, Vermont uses "acceptable management practice;" Connecticut uses "guidelines and suggestions;" and California uses "forest practice rule."

The Water Quality Act of 1987 further amended the Clean Water Act, encouraging implementation of BMPs by requiring planning procedures that made the link between cause and effect more explicit. This was accomplished by requiring (1) detailed water quality plans that identified water bodies not meeting water quality standards, (2) identification of categories of nonpoint sources or particular nonpoint sources responsible for violation of water quality standards in those water bodies, and (3) identification of BMPs to control them.

Continuing concern about nonpoint source pollution, and associated federal, state, and local legislation, encouraged more proactive state efforts to protect water quality on forest lands. In the past 4 years, several more states adopted BMPs for forest lands, and many states with programs increased their efforts to have their BMPs understood and used. State programs and federal agency efforts, surveyed during the spring of 1992, are summarized here. For more detail, see Brown et al. (1993).

Type of Program

State approaches can be broadly categorized as regulatory or voluntary. Regulatory programs impose requirements on land management and allow assessment of fines and other penalties for noncompliance. States with regulatory programs usually rely on inspection of management activities while the activities are in progress, as well as follow-up inspections, to improve compliance with BMPs and to determine whether penalties are to be assessed. Regulatory states also may require approval of harvesting or road construction plans that include water quality protection measures before field work begins. States with voluntary programs emphasize education and training (BMP manuals, seminars, mailings and personal contacts), and onsite inspection, where requested. On forest land across the 50 states, in 1992, 23 states had voluntary programs, 13 had regulatory programs, 5 others used a combination of voluntary and regulatory measures, and 9 still lacked a formal program.

In addition to state programs, many local ordinances have been passed by counties, townships, and municipalities. Martus et al. (1993) identified 522 local ordinances that regulate forestry activities in the U.S. Half are written solely for forestry purposes, and half are written broadly enough to be interpreted as affecting forestry activities. These ordinances were found in 24 states, with 68% of them in northeastern states and 27% in southern states. Local ordinances are less common in the West, where more compre-

hensive state-level laws are more common. About three-quarters of the ordinances were enacted in the past 10 years, and nearly half are less than 5 years old.

Financial Incentives

Eleven states now offer state-funded cost sharing and or tax incentive programs to encourage private land owners to use forest and rangeland BMPs or implement other activities that will have a positive effect on water quality (Brown et al. 1993). These programs focus on a variety of water quality protection measures, including use of BMPs in harvest and related road construction, reforestation, protection of riparian areas from damage by livestock, and maintenance of woodlands.

Four federally funded programs provide cost-share funds for forestry activities, on forest or agricultural land, that may protect or improve water quality. Funds for these programs are disbursed by the Agricultural Stabilization and Conservation Service; but forestry aspects of the programs are facilitated by the U.S. Forest Service, in cooperation with state personnel. The Agricultural Conservation Program, begun in 1936, supports a series of agricultural conservation practices including tree planting and stand improvement; more than 7 million acres have been planted so far, mainly in the southern states. The Conservation Reserve Program, established in 1985 and expected to end in 1995, funds actions to retire farm land by establishing permanent cover; more than 2.3 million acres have been planted with trees in 41 states, with 92% in the southern states. The Forestry Incentive Program, established in 1974 and slated to end in 1995, funds timber production activities, including tree planting and stand improvement; more than 3.9 million acres in 49 states have benefitted so far, with 70% in the southern states. Finally, the Stewardship Incentive Program, which began disbursing funds in 1992, supports environmental protection activities, including streambank stabilization, riparian buffer zones, and protection of native vegetation.

Implementation Monitoring

Even if BMPs are appropriately specified for the site, they must be implemented. States use different procedures to encourage BMP implementation and to estimate compliance with their nonpoint source pollution programs on forest lands. Eleven states reported that agency personnel visit some or all sites when land disturbance activities are in progress, to monitor compliance. This approach is common in states with regulatory programs. Because ongoing inspection of forest management in progress is expensive, inspectors may visit only the most important sites. Twenty-two states reported performing formal post-hoc surveys of all or randomly selected, recently managed (e.g., harvested) sites. This approach is common in states with voluntary programs. Many of these 22 states only recently began to formally measure the degree of BMP use. Besides states performing formal surveys, 5 states reported doing post-hoc inspections of miscellaneous sites selected by field foresters; and 8 states reported performing inspections based on complaints or other knowledge of problems. In all, 40 states reported using some procedures for monitoring compliance, with some using more than one procedure. BMP monitoring also occurs where contracts between the state and a private party require BMP use. This occurs where land owners benefit from cost sharing or tax incentive plans, and where contracts for harvest on state land require use of BMPs (which may happen even though the state lacks a formal BMP program).

Most states performing formal implementation surveys reported overall (across all ownerships and BMPs) compliance of at least 85%, and usually above 90%. However, compliance differed by ownership and type of BMP. In general, compliance tended to be lower for private than public or industry land, and lower for small land owners than large land owners. States differ in which BMPs are more, and less, often complied with; but as a general rule, road and skid trail BMPs had the lower compliance rates. In voluntary states, compliance with some BMPs on private land fell below 50%. In states with well-established regulatory programs, the most common problems were with failure to properly fill out reports or notify state officials; compliance with practices on the ground tended to be at least 95%.

The clear trend among the states is towards a more concerted monitoring effort, using periodic surveys with well-established survey methods. The number of states performing formal surveys of BMP compliance has increased dramatically; and formal surveys of randomly selected sites appears to be the preferred approach. Encouraging results from such surveys

generally are considered to be necessary justification for not switching from a voluntary to a regulatory nonpoint source pollution control program.

Effectiveness Monitoring

Effectiveness of BMP use can be checked in two ways: qualitatively by trained professionals during onsite inspection, or by quantitative measurement. Qualitative checking can be informal, or, preferably, via a formal survey of randomly selected sites, perhaps together with an implementation survey. Quantitative measurement can include water quality sampling, bedload monitoring, and biological monitoring, as well as on-land monitoring of soil movement.

Twelve states reported that they now perform formal periodic post-hoc qualitative surveys of BMP effectiveness at selected sites. Formal surveys are more common in states with regulatory BMP programs. In addition, seven other states reported performing qualitative effectiveness inspections, usually in response to complaints or during other site inspections. Only five states reported performing quantitative water quality measurements to evaluate BMP effectiveness. In all, 21 states reported performing some type of effectiveness monitoring. The difficulty of adequately measuring effectiveness with qualitative methods and the high costs of performing sufficient water quality sampling to evaluate BMP effectiveness both contribute to the lack of state efforts in this area.

The qualitative assessments of BMP effectiveness reviewed suggested that forest practices generally can avoid significant deterioration of water quality if BMPs are carefully developed and used. Most current water quality problems associated with forest practices probably result from poor implementation of BMPs. However, some qualifications to this are needed: (1) it is not clear if retention of vegetated buffers along streams will reduce peak nitrate concentrations in all areas; (2) most state BMPs do not carefully protect ephemeral channels from disturbance, although such channels may produce or deliver much of the sediment reaching fish-bearing streams; (3) BMPs to minimize mass movements on unstable slopes are still in the design phase, in most states; and (4) use of BMPs to avoid erosion from livestock grazing in riparian areas is lacking.

Federal Land Management Agency Nonpoint Source Pollution Control Efforts

According to the Clean Water Act, federal lands must comply with state water quality laws and standards to the same extent as any nongovernmental entity. Although states may monitor BMP implementation and effectiveness on federal land, they typically rely on federal agencies to monitor on federal lands and focus state resources on state and private lands.

There has been much recent activity by federal agencies to respond to the Clean Water Act goals; and many regional offices without carefully developed monitoring plans are in the process of developing them. Federal BMP monitoring programs usually are established to comply with state BMP monitoring programs developed using section 319 of the Clean Water Act. These programs attempt to maintain a balance between a decentralized approach (e.g., sampling frequency is defined at the district or forest level) and a more centralized approach (e.g., sampling frequency is defined at the regional level). The advantage of a decentralized approach is that monitoring can be tailored to individual land use activities. The disadvantage of this approach is that monitoring activities vary across space and time, and data cannot be easily compared or aggregated to the forest or regional levels.

Cost Effectiveness of BMPs

The goal of water quality protection programs is to meet water quality standards most cost-effectively. BMPs are an administrative approach to reaching this goal. Specifying BMPs to cost-effectively reach water quality standards requires an understanding of the complex relationships between land disturbance and downstream water quality, as well as the costs of alternative BMPs. Water quality monitoring is essential to understand the relationship between land disturbance and water quality. By observing the effect, over time, of precipitation on water quality downstream of disturbed and undisturbed areas, scientists and land managers improve their understanding of these relationships. This improved understanding then can be used to reassess BMP guidelines to more cost-effectively reach water quality goals. This iterative process of BMP specification,

use, monitoring, and then fine-tuning of BMP specifications for future applications is the key to cost-effective BMP use and effective water quality protection. This relies heavily on gradually improved understanding of the effect of site-specific land management controls on downstream water quality.

Water quality standards are cost-effectively met when they neither over- nor under-constrain land management. The cost of over-constraining land management is in the wasted resources and lost income for the land owners. The cost of under-constraining land management is the effect of poor water quality on aquatic organisms and downstream water users, and the site productivity loss. The more carefully BMPs are tailored to site-specific conditions, the more likely it is that they will cost-effectively reach their stated goals. Because the professional expertise to carefully select BMPs is costly, BMPs sometimes are specified for large geographical areas (such as counties or multi-county regions), although nonpoint source pollution at specific sites within the larger area may be more inexpensively controlled with one set of BMPs than another. This is not the fault of the BMP approach; rather, it is a matter of how BMPs are specified. The availability of well-qualified personnel at the field level is probably the most cost-effective approach to meeting water quality standards.

Summary and Conclusions

The quality of water draining forested watersheds is typically the best in the Nation, whether the forests are left untouched or managed. Water quality problems on forest land are highly variable over space and time. Relatively few forest areas of the country, if carefully managed, are prone to troublesome pollutant yields. However, forest practices sometimes are poorly implemented , leading to degradation of water quality. Further, some past practices caused impacts that will take decades to work through the system.

Sediment loads in streams is the most widespread water pollution problem in forests. Sediment concentrations may drastically exceed water quality objectives, even from undisturbed watersheds during rare, intense storms; and land disturbance by forest management may significantly increase those concentrations. Roads are a major contributor to sediment concentrations in streams; road design and maintenance are critical to minimize sediment problems. Some silvicultural practices also can significantly increase sediment concentrations in areas with sensitive soils. The ecological impacts of increased sediment production from forest practices have received the greatest attention in the Pacific Northwest, where high rainfall, steep slopes, erodible soils, and valuable fisheries combine to accentuate the problem. Impacts of forest practices on sediment yields have been as great in other regions; but information about fish habitat impacts is lacking. Few studies have directly assessed impacts of grazing on water quality. Management practices that protect streambanks and riparian vegetation are likely to protect water quality from substantial grazing impacts.

Other categories of water quality degradation on forests and rangeland are much less serious than sedimentation. Forest practices generally have little impact on oxygen levels or on dissolved solids. Pathogen problems can be controlled by protecting riparian areas from grazing and by providing facilities for recreationists or limiting use rates in high use areas. Nitrate generally is the only ion of critical interest in relation to forest practices. Harvesting markedly increases nitrate concentrations in the chaparral and northern hardwood areas, and application of nitrogen fertilizers also may cause stream nitrate levels to peak at high concentrations. Herbicide applications that follow regulatory guidelines have not impaired water quality (Norris et al. 1991). Temperature increases caused by harvesting harm aquatic life. Retention of buffer strips appears to be an effective approach to avoiding harmful stream temperature increases, and also slows the movement of sediment towards streams, although the amount of buffer needed for different conditions is not well understood.

The large variety of state and federal agency programs and procedures to protect water quality on forests and rangelands makes concise summary impossible. Furthermore, merely stating that some program or approach is used provides little indication of the degree to which it is funded or the energy with which it is administered. Nevertheless, the many additions to such programs in the past few years indicate that water quality protection on forest lands is being taken more seriously. Also, protection on rangelands is beginning to receive some formal attention

At least 40 states use some procedures to monitor BMP implementation on forest lands. In states with well-established regulatory programs, compliance with practices on the ground tends to be at least 95%. Most states with voluntary programs that performed formal BMP implementation surveys reported overall compliance of at least 85%, and usually above 90%. However, compliance with some BMPs, and compliance by some classes of land owners, falls considerably below these levels. In general, compliance is lower for private than public or industry land, and lower for small land holdings than for large land holdings.

Because most, if not all, of the onsite costs of BMP implementation are borne by the landowner, while the benefits typically accrue to aquatic organisms and downstream water users, landowners sometimes may view noncompliance as an preferable alternative, especially in voluntary states. Therefore, monitoring must be an ongoing activity; and voluntary states may need to seriously consider instituting a regulatory program. Enhanced education, technical assistance, and monitoring are needed for nonindustrial private forest lands. States not periodically monitoring BMP implementation on a set of randomly selected sites should be encouraged to do so.

Effectiveness monitoring is less common than implementation monitoring, largely because of the difficulty of measuring effectiveness with qualitative methods and the high cost of performing the level of water quality sampling that would be required. Effectiveness monitoring efforts indicate that implementation of current BMP specifications avoids most deleterious effects of forest and rangeland management practices. Research is needed to improve methods for monitoring the effectiveness of BMPs.

The cost-effectiveness with which BMPs meet quality standards depends, in part, on how well the BMPs are tailored to specific conditions at the site. Generally, the availability of well-qualified personnel at the field level to provide site-specific BMP recommendations is probably the most efficient way to meet water quality standards on forests and rangeland.

References

Aber, J.D.; Nadelhoffer, K.J.; Steudler, P.; Melillo, J.M. 1990. Nitrogen saturation in northern forest ecosystems. BioScience. 39: 378-386.

Askew, G.R.; Williams, T.M. 1986. Water quality changes due to site conversion in coastal South Carolina. Southern Journal of Applied Forestry. 10: 134-136.

Aubertin, G.M.; Patric, J.H. 1974. Water quality after clearcutting a small watershed in West Virginia. Journal of Environmental Quality. 3: 243-249.

Bateridge, T. 1974. Effects of clearcutting on water discharge and nutrient loss, Bitterroot National Forest, Montana. M.S. thesis, Office of Water Resources, University of Montana, Missoula.

Beasley, R.S. 1979. Intensive site preparation and sediment losses on steep watersheds in the Gulf Coastal Plain. Soil Science Society of America Journal. 43: 412-417.

Beschta, R.L. 1990. Effects of fire on water quantity and quality. In: J.D. Wlastad, S.R. Radusevich, and D.V. Sandberg, eds. Natural and prescribed fire in Pacific Northwest Forests. Corvallis, OR: Oregon State University Press: 219-231.

Biesecker, J.E.; Leifeste, D.K. 1975. Water quality of hydrologic bench marks—an indicator of water quality in the natural environment. U.S. Geological Survey Geological Survey Circular 460-E, Reston, VA.

Bigley, R.E.; Kimmins, J.P. 1983. Herbicide effects on ecosystem nitrogen loss. In: Proceedings of the 1983 National Convention of the Society of American Foresters.

Binkley, D.; Brown, T.C. 1993a. Forest practices as nonpoint sources of pollution in North America. Water Resources Bulletin. 29(5): 729-740.

Binkley, D.; Brown, T.C. 1993b. Management impacts on water quality of forests and rangelands. Gen. Tech. Rep. RM-239 Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 114 p.

Bisson, P.A. 1982. Annual fertilization and water quality—final report. Weyerhaeuser Company, Technical Report 050-5411-02, Centralia, WA.

Bisson, P.A. 1988. Final report: 1988 Weyerhaeuser forest fertilization/water quality monitoring studies: Forks Creek, Ryderwood, Silver Lake. Tacoma, WA: Weyerhaeuser Company Technology Center.

Bisson, P.A.; Ice, G.G.; Perrin, C; Bilby, R. 1992. Effects of forest fertilization on water quality and aquatic resources in the Douglas-fir region. In: H.N. Chappell, G.F. Weetman, and R.E. Miller, eds. Forest fertilization: sustaining and improving growth of western forests. Seattle, WA: University of Washington, Institute of Forest Resources, Contribution: 72: 179-193.

Black, Peter E. 1992. Reauthorization of the clean wate act. Water Resources Update issue 88 pages 5-6, Carbondale, FL: Universities Council on Water Resources.

Blackburn, W.H.; Wood, J.C.; DeHaven, M.G. 1986. Storm flow and sediment losses from site-prepared forestland in east Texas. Water Resources Research. 22: 776-784.

Brown, G.W. 1989. Forestry and water quality. Corvallis, OR: Oregon State University Bookstore.

Brown, G.W.; Krygier, J.T. 1970. Clearcut logging and sediment production in the Oregon Coast Range. Water Resources Research. 7: 1189-1198.

Brown, H.E.; Baker, M.B.; Rogers, J.J.; Clary, W.P.; Kovner, J.L.; Larson, F.R.; Avery, C.C.; Campbell, R.E. 1974. Opportunities for increasing water yields and other multiple use values on ponderosa pine forest lands.

Brown, T.C.; Brown, D.; Binkley, D. 1993. Laws and programs for controlling nonpoint source pollution in forest areas. Water Resources Bulletin. 29(1):1-13.

Brown, T.C.; Harding, B.L.; Payton, E.A. 1990. Marginal economic value of streamflow: a case study for the Colorado River Basin. Water Resources Research. 26: 2845-2859.

Buckhouse, J.; Gifford, G. 1976. Water quality implementations of cattle grazing on a semiarid watershed in southeastern Utah. Journal of Range Management. 29: 109-113.

Cederholm, C.; Reid, L. 1987. Impact of forest management on coho salmon (Oncorhynchus kisutch) populations of the Clearwater River, Washington: a project summary. In: E.O. Salo and T.W. Cundy, eds. Streamside Management: Forestry and Fishery Interactions Seattle, WA: Institute of Forest Resources Contribution #57, University of Washington: 373-398.

Chaney, E.; Elmore, W.; Platts, W.S. 1990. Livestock grazing on western riparian areas. Produced for the U.S. Environmental Protection Agency. Wahsington, DC: U.S. Government Printing Office.

Clayton, J.L.; Kennedy, D.A. 1985. Nutrient losses from timber harvest in the Idaho batholith. Soil Science Society of America Journal. 49: 1041-1049.

Curry, R.R. 1987. Water quality protection in forest management: are best management practices working? In: R.Z. Callaham and J.J. DeVries, technical coordinators. California Watershed Management Conference Wildland Resources Center Technical Report #11. Berkeley, CA: University of California: 55-61.

Darling, R.A.; Coltharp, G.B. 1973. Effects of livestock grazing on the water quality of mountain streams. In: Proceedings of the Symposium on Water-Animal Relations. Kimberly, ID: Water-Animal Relations Committee, U.S. Department of Agriculture, Agricultural Research Service: 1-8.

Davis, E.A. 1987a. Chaparral conversion to increase streamflow in Arizona: sequential treatments extend duration of nitrate loss to streamwater. Forest Science. 33: 89-103.

Edwards, P.J.; Kochenderfer, J.N.; Seegrist, D.W. 1991. Effects of forest fertilization on stream water chemistry in the Appalachians. Water Resources Bulletin. 27: 265-274.

EPA. 1978. Microbiological methods for monitoring the environment. EPA-600/8-78-017, Cincinatti, Ohio.

EPA. 1980. An approach to water resources evaluation of non-point silvicultural sources. EPA-600/8-80-012, Washington, DC.

EPA. 1986. Quality criteria for water: U.S.E.P.A. Office of Water Regulations and Standards. Washington D.C.

EPA. 1990. National water quality inventory: 1988 report to Congress. EPA 440-4-90-003, Office of Water, U.S. Environmental Protection Agency, Washington, D.C.

ERS. 1989. Agricultural resources: cropland, water, and conservation. Situation and outlook report. AR-16, U.S. Department of Agriculture, Economic Research Service, Washington, D.C.

Everest, F.H.; Beschta, R.L.; Scriverner, J.C.; Koski, K.V.; Sedell, J.R; Cederholm, C.F. 1987. Fine sediment and salmonid production: a paradox. In: E.O. Salo and T.W. Cundy, eds. Streamside Management: Forestry and Fishery Interactions Seattle, WA: College of Forest Resources Contribution #57, University of Washington: 98-142.

Feller, M.C. 1981. Effects of clearcutting and slashburning on stream temperature in southwestern British Columbia. Water Resources Bulletin. 17: 863-867.

Feller, M.C.; Kimmins, J.P. 1984. Effects of clearcutting ansd slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. Water Resources Research. 20: 29-40.

Ficke, J.F.; Hawkinson, R.O. 1975. The national stream quality accounting network (NASQAN) - some questions and answers. Circular 719, U.S. Geological Survey, Reston, Virginia.

Fowler, W.B.; Anderson, T.D.; Helvey, J.D. 1988. Changes in water quality and climate after forest harvest in central Washington state. Res. Pap. PNW- 388. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest research Station.

Fredriksen, R.; Moore, D.; Norris, L. 1975. The impact of timber harvest, fertilization, and herbicide treatment on streamwater quality in western Oregon and Washington. In: B. Bernier and C. Winget, eds. Forest Soils and Land Management. Quebec, Canada: Laval University Press: 283-313.

Fredriksen, R.L.; Harr, R.D. 1979. Soil, vegetation, and watershed management. In: Heilman, P.E. ;Anderson, H.W.; Baumgartner, D.M., eds. Forest Soils of the Douglas-fir . Pullman, WA: Washington State University: 231-260.

Gianessi, L.P.; Peskin, H.M.; Puffer, C.A. 1986. National data base of nonurban-nonpoint-source discharges and their effect on the nation's water quality. Prepared for the U.S. Environmental Protection Agency by Resources for the Future, Washington, D.C.

Hall, J.; Lantz, R. 1969. Effects of logging on the habitat of coho salmon and cutthroat trout in coastal streams. In: A Symposium on Salmon and Trout (T. Northcote, ed.). Vancouver, Canada: University of British Columbia.

Hartman, G.; Scrivener, J.; Holtby, L.; Powell, L. 1987. Some effects of different streamside treatments on physical conditions and fish populations processes in Carnation Creek, a coastal rain forest stream in British Columbia. In: E.O. Salo and T.W. Cundy, eds. Streamside Management: Forestry and Fishery Interactions. Seattle, WA: Institute of Forest Resources Contribution #57, University of Washington: 330-372.

Helvey, J.D.; Kochenderfer, J.N.; Edwards, P.J. 1989. Effects of forest fertilization on selected ion concentrations in central Appalachian streams. In: Proceedings of the seventh central hardwood conference. Gen. Tech. Rep. NC-132. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Foresest Experiment Station. 278-282 p.

Hem, J.D. 1989. Study and interpretation of the chemical characteristics of natural water. Water Supply Paper 2254. Reston, VA: U.S. Geological Survey.

Hetherington, E.D. 1976. Dennis Creek: a look at water quality following logging in the Okanagan Basin. Environment Canada, Canadian Forestry Service.

Hetherington, E.D. 1985. Streamflow nitrogen loss following fertilization in a southern Vancouver Island watershed. Canadian Journal of Forest Research. 15: 34-41.

Hewlett, J.D.; Fortson, J.C. 1982. Stream temperature under an inadequate buffer strip in the southeastern Piedmont. Water Resources Bulletin. 18: 983-988.

Hewlett, J.D.; Post, H.E.; Doss, R. 1984. Effect of clearcut silviculture on dissolved ion export and water yield in the Piedmont. Water Resources Research. 20: 1030-1038.

Holtby, L.B. 1988a. The effects of logging on stream temperatures at Carnation Creek. In: Chamberlin, T.W., ed. Proceedings of the Workshop: Applying 15 years of Carnation Creek Results Nanaimo, British Columbia, Canada: Pacific Biological Station: 118-122.

Holtby, L.B. 1989b. The effects of logging on the coho salmon of Carnation Creek, British Columbia. In: Proceedings of the Workshop: Applying 15 years of Carnation Creek Results (T.W. Chamberlin, ed.) Nanaimo, British Columbia, Canada: Pacific Biological Station: 159-174.

Hornbeck, J.; Corbett, E.; Duffy, P.; Lynch, J. 1984. Forest hydrology and watershed management. In: K. Wenger, ed. Forestry Handbook, Wiley, New York: 637-678.

Hornbeck, J.W.; Martin, C.W.; Pierce, R.S.; Bormann, F.H.; Likens, G.E.; Eaton, J.S. 1987. The northern hardwood forest ecosystem: ten years of recovery from clearcutting. Res. Pap. NE-596. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeast Forest Experiment Station.

Ice, G.G. 1978. Reaeration in a turbulent stream system. Corvallis: Oregon State University. PhD thesis.

Johnson, D.; Lindberg, S. 1992. Atmospheric deposition and nutrient cycling in forest ecosystems. Springer-Verlag, New York.

Johnson, D.W. 1991. Nitrogen retention in forest soils. Journal of Environmental Quality.

Judy, R.D., Jr.; Seeley, P.N.; Murray, T.M.; Svirsky, S.C.; Whitworth, M.R.; Ischinger, L.S. 1984. 1982 national fisheries survey. Volume I technical report: initial findings. FWS/OBS-84/06, U.S. Fish and Wildlife Service: Washington, D.C.

Kochenderfer, J.N.; Aubertin, G.M. 1975. Effects of management practices on water quality and quantity: In Proceedings of Symposium on Municipal Watershed Management: Gen. Tech. Rep. NE-13. Broomall, PA: U.S. Department of Agriculture, Forest Service, Northeast Forest Experiment Station. 14-24 .

Krause, H.H. 1982. Nitrate formation and movement before and after clear-cutting of a monitored watershed in central New Brunswick, Canada. Canadian Journal of Forest Research. 12: 922-930.

Lettenmaier, D.P.; Hooper, E.R.; Wagoner, C.; Faris, K.B. 1991. Trends in stream quality in the continental Unites States, 1978-1987. Water Resources Research. 27: 327-339.

Levno, A.; Rothacher, J. 1969. Increases in maximum stream temperatures after slash burning in a small experimental watershed. Res. Note PNW-110. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Experiment Station.

Likens, G.; Bormann, F.H.; Johnson, N.; Fisher, D.; Pierce, R. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecological Monographs. 40: 23-47.

Lynch, J.A.; Corbett, E.S. 1990. Evaluation of best management practices for controlling nonpoint pollution from silvicultural operations. Water Resources Bulletin. 26: 41-52.

Lynch, J.A.; Sopper, W.E.; Corbett, E.S.; Aurand, D.W. 1975. Penn State experimental watersheds. pp. 32-41 In: Municipal Watershed Management. Gen. Tech. Rep. NE-13. Upper Darby, PA: U.S. Department of Agriculture, Forest Service, Northeast Forest Experiment Station.

MacDonald, L.; Smart, A.; Wismar, R. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. USEPA/910/9-91-001, Region 10, Seattle.

Martin, C.W.; Noel, D.S.; Federer, C.A. 1984. Effects of forest clearcutting in New England on stream chemistry. Journal of Environmental Quality. 13: 204-210.

Martus, C.E.; Haney, H.L., Jr,; Siegel, W.C. 1993. The impact of local forestry-related ordinances in the Unites States. Proceedings of Southern Forest Economics Workers' Annual Workshop, Research Triangle Park, NC.

McClurkin, D.C.; Duffy, P.D.; Ursic, S.J.; Nelson, N.S. 1985. Water quality effects of clearcutting upper Coastal Plain loblolly pine plantations. Journal of Environmental Quality. 14: 329-332.

Meehan, W.R.; Lotspeich, F.B.; Mueller, E.W. 1975. Effects of forest fertilization on two southeast Alaska streams. Journal of Environmental Quality. 4: 50-55.

Miller, J. 1974. Nutrient losses and nitrogen mineralization on forested watersheds in Oregon's Coast Range. Corvallis: Oregon State University, Corvallis: PhD thesis.

Miller, R.; Fight, R. 1979. Fertilizing Douglas-fir forests. Gen. Tech. Rep. PNW-83. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Experiment Station.

Miller, J.H.; Newton, M. 1983. Nutrient losses from disturbed forest watersheds in Oregon's Coast Range. Agro-Ecosystems. 8: 153-167.

Moore, E.; Janes, E.; Kinsinger, F.; Pitney, K.; Sainsbury, J. 1979. Livestock grazing management and water quality protection (state of the art reference document). EPA 910/9-79-67, Seattle, WA.

National Water Commission. 1973. Final report of the National Water Commission, water policies for the future. Washington, D.C.

Newton, M.; Norgren, J.A. 1977. Silvicultural chemicals and protection of water quality. USEPA Report #910/0-77-036, Seattle, WA.

Noble, E. 1965. Sediment reduction through watershed rehabilitation. Page 114-123 in Proceedings of the Federal Interagency Sedimentation Conference, Washington, DC: USDA Misc. Publ. #970.

Norris, L.A.; Lorz, H.W.; Gregory, S.V. 1991. Forest chemicals. In: William R. Meehan, ed. Influences of forest and rangeland management on salmonid fishes and their habitats. Bethesda, MD: American Fisheries Society Special Publication 19: 207-296.

Omernik, J.M. 1977. Nonpoint source - stream nutrient level relationships: a nationwide study. EPA-600/3-77-105, Environmental Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Corvallis, Oregon.

Patric, J.H. 1980. Effects of wood products harvest on forest soil and water relations. Journal of Environmental Quality. 9: 73-80.

Peters, N.E. 1984. Evaluation of environmental factors affecting yields of major dissolved ions of streams in the United States. Water Supply Paper 2228. Reston, VA: U.S. Geological Survey.

Plamondon, A.P.; Gonzalez, A.; Thomassin, Y. 1982. Effects of logging on water quality: comparison between two Quebec sites. In: Canadian Hydrology Symposium #82. Fredrikton, Canada: University of New Brunswick: 49-70.

Potter, L.D.; Gosz, J.R.; Carlson, C.A. 1984. Water resources in the Southern Rockies and High Plains: forest recreational use and aquatic life. Albuquerque, NM: University of New Mexico Press.

Pye, J.; Vitousek, P. 1985. Soil and nutrient removals by erosion and windrowing at a Southeastern U.S. Piedmont site. Forest Ecology and Management. 11: 145-155.

Rice, R.M.; Tilley, F.B.; Datzman, P.A. 1979. A watershed's response to logging and roads: South Fork of Caspar Creek, California, 1967-1976. Res. Pap. PSW-146. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station.

Richter, D.; Ralston, C.W.; Harms, W.R. 1982. Prescribed fire: effects on water quality and forest nutrient cycling. Science. 215: 661-663.

Riekirk, H. 1983. Impacts of silviculture on flatwoods runoff, water quality, and nutrient budgets. Water Resources Bulletin. 19: 73-79.

Riggan, P.J.; Lockwood, R.N.; Lopez, E.N. 1985. Deposition and processing of airborne nitrogen pollutants in Mediterranean-type ecosystems of southern California. Environmental Science and Technology. 19: 781-789.

Rishel, G.B.; Lynch, J.A.; Corbett, E.S. 1982. Seasonal stream temperature changes following forest harvest. Journal of Environmental Quality. 11: 112-116.

Salminen, E.M.; Beschta, R.L. 1991. Phosphorus and forest streams: the effects of environmental conditions and management activities. Corvallis, OR: Department of Forest Engineering, Oregon State University.

SCS. 1984. National resources inventory: a guide for users of 1982 NRI data files. Soil Conservation Service, U.S. Department of Agriculture, Statistical Laboratory, Ames, IA: Iowa State University.

Silsbee, D.; Larson, G. 1982. Water quality of streams in the Great Smoky Mountains National Park. Hyrobiologia. 89: 97-115.

Smith, R.A.; Alexander, R.B.; Wolman, M.G. 1987. Water quality trends in the nation's rivers. Science. 235: 1607-1615.

Snyder, G.G.; Haupt, H.F.; Belt, G.H., Jr. 1975. Clearcutting and burning slash alter quality of stream water in northern Idaho. Research Pap. INT-168. Ogden, UT: U.S. Department of Agriculture, Forest Service.

Sollins, P.; McCorison, F.M. 1981. Nitrogen and carbon solution chemistry of an old growth coniferous forest watershed before and after cutting. Water Resources Research. 17: 1409-1418.

Stednick, J.D.; Tripp, L.N.; McDonald, R.J. 1982. Slash burning effects on soil and water chemistry in southeastern Alaska. Journal of Soil and Water Conservation. 37: 126-128.

Stottlemyer, R. 1987. Natural and anthropic factors as determinants of long-term streamwater chemistry. In: C.A. Troendle, M.R. Kaufmann, R.H. Hamre, and R.P. Winokur, eds. Management of subalpine forests: building on 50 years of research. Gen. Tech. Rep. RM-149, Fort Collins, CO: U.S.Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 86-94 p.

Sullivan, K. 1985. Long-term patterns of water quality in a managed watershed in Oregon: 1. Suspended sediment. Water Resources Bulletin. 21: 977-987.

Sullivan, K.; Lisle, T.E.; Dolloff, C.A.; Grant, G.E.; Reid, L.M. 1987. Stream channels: the link between forests and fishes. In: E.O. Salo and T.W. Cundy, eds. Streamside management: forestry and fishery interactions. Seattle, WA: College of Forest Resources Contribution 57, University of Washington: 39-97.

Sullivan, K.; Tooley, J.; Doughty, K.; Caldwell, J.E.; Knudson, P. 1990. Evaluation of prediction models and characterization of stream temperature regimes in Washington. Timber/Fish/Wildlife Rep. No. TFW-WQ3-90-006. Olympia, WA: Washingtion Department of Natural Resources.

Swank, W.T. 1988. Stream chemistry responses to disturbance. In: W.T. Swank and D.A. Crossley, eds. New York, NY: Forest Hydrology and Ecology at Coweeta Springer Verlag: 339-358.

Swanson, F.J.; Dyrness, C.T. 1975. Impact of clearcutting and road construction on soil erosion by landslides in the western Cascade Range, Oregon. Geology. 3: 393-396.

Swanston, D.N. 1991. Natural Processes. Chapter 5 in Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. American Fisheries Society Special Publication 19: 139-179.

Swanston, D.N.; Swanson, F.J. 1976. Timber harvesting, mass erosion, and steepland forest geomorphology in the Pacific Northwest. In: D. Coates, ed. Geomorphology and Engineering Dowden, Hutchinson and Ross, Stroudsberg, Pennsylvania: 199-221.

Swift, L.W., Jr. Messer, J.B. 1971. Forest cuttings raise temperatures of small streams in the southern Appalachians. Journal of Soil and Water Conservation. 26: 111-116.

Tiedemann, A.R.; Quigley, T.M.; Anderson, T.D. 1988. Effects of timber harvest on stream chemistry and dissolved nutrient losses in northeast Oregon. Forest Science. 2: 344-358.

Tiedemann, A.; Helvey, J.; Anderson, T. 1978. Stream chemistry and watershed nutrient economy following wildfire and fertilization in eastern Washington. Journal of Environmental Quality. 7: 580-588.

U.S. Water Resources Council. 1978. The Nation's Water Resources 1975-2000. Second National Assessment by the U.S. Water Resources Council. U.S. Government Printing Office, Stock #052-00082-7, Washington, D.C.

U.S. Forest Service. 1983. Land areas of the National Forest System as of September 30, 1983. FS-383, Forest Service, USDA, Washington, D.C.

Van Lear, D.H.; Douglas, J.E.; Cox, S.K.; Augspurger, M.K. 1985. Sediment and nutrient export in runoff from burned and harvested pine watersheds in the South Carolina Piedmont. Journal of Environmental Quality. 14: 169-174.

Verry, E.S. 1972. Effect of an aspen clearcutting on water yield and quality in northern Minnesota. In: S.C. Csallany, T.G. McLaughlin, and W.D. Striffler, eds. National Symposium on Watersheds in Transition American Water Resources Association, Urbana, IL: 276-284.

Walstad, J.; Kuch, P. 1987. Forest vegetation management·for conifer production. Wiley, New York.

Welsch, D.J. 1992. Riparian forest buffers: function and design for protection and enhancement of water resources. Radnor, PA: USDA Forest Service, State and Private Forestry, NA-PR-07-91.

Abstract

Brown, Thomas C.; Binkley, Dan. 1994. Effect of management on water quality in North American forests. General Technical Report RM-248. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 27 p.

Although the quality of water draining forested watersheds is typically the best in the Nation, some forest management practices can seriously impair streamwater quality. Sediment is the main concern. High suspended sediment levels, and adverse changes in stream channels, are potential problems in several regions, especially after road construction, and some harvesting and grazing practices. Impacts are most serious where fish reproduction is affected. Nitrate and water temperature are less serious problems. Harvesting can increase nitrate levels markedly, in some locations; and removal of overstory from along streambanks can raise water temperatures enough to impair fish survival. Best management practices (BMPs) can avoid most of these harmful effects. Additional work is needed, in some locations, to encourage BMP use and to tailor BMP specifications to site-specific conditions.

Keywords: Water quality, forest management, RPA, assessment